Nitrogen Dynamics at a Manured Grass Field Overlying the Sumas-Blaine Aquifer in Whatcom County

March 2014

Publication No. 14-03-001

DEPARTMENT OF
ECOLOGY
State of Washington

Publication and contact information

This report is available on the Department of Ecology's website at
https://fortress.wa.gov/ecy/publications/SummaryPages/1403001.html

The following supplementary documents are linked to this report at the website above:

- Appendices B through T
- Response to Comments

Data for this project are available at Ecology's Environmental Information Management (EIM) website www.ecy.wa.gov/eim/index.htm. Search Study ID, bcar0003.

Ecology's Study Tracker Code for this study is 02-039.

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Cover photo: Liquid manure application (photo by Barbara Carey).

Nitrogen Dynamics at a Manured Grass Field Overlying the Sumas-Blaine Aquifer in Whatcom County

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Water Resource Inventory Area (WRIA) and 8-digit Hydrologic Unit Code (HUC) numbers for the study area:

- WRIA: 1 - Nooksack
- HUC number: 17110004

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Abstract

The Washington State Department of Ecology, in cooperation with Washington State University, conducted a 4-1/2-year intensive monitoring study at a manured grass field overlying the Sumas-Blaine Aquifer in northwest Washington. The purpose of the study was to evaluate nitrogen dynamics in dairy manure, soil, crop, and groundwater at a field overlying the Sumas-Blaine Aquifer.

We quantified the mass of nitrogen added to the field in the form of manure, inorganic fertilizer, and irrigation water; the mass of nitrogen removed in the crop; the mass of nitrate in the soil during the post-harvest period; and the concentration of nitrate in groundwater beneath the field. Shallow depth to water (0 to 11 feet) enabled rapid responses to nitrate transport, especially during the high-rainfall period (October through March).

Average monthly nitrate concentrations in 6 shallow monitoring wells ranged from 5.5 to 30 mg/L-N with a maximum in one well of 45 mg/L-N. Early winter average nitrate concentrations in groundwater, representing newly recharged water carrying nitrate from the soil column, were (1) above 10 mg/L-N following growing seasons with nitrogen loading greater than the mass of nitrogen removed in the crop and (2) generally below 10 mg/L-N when nitrogen loading was similar to crop removal. Other factors that affected nitrate concentrations in groundwater included timing of manure applications, tillage of the field shortly before the study began, and denitrification in the aquifer.

Model results based on measured field parameters indicated an average of 115 lb/acre of nitrate leached to groundwater from September through March. Two methods for estimating the nitrogen residual at the end of the growing season, mass balance analysis and post-harvest soil nitrate testing, were not reliable predictors of nitrate concentrations in groundwater. Direct monitoring of water quality at the water table was the only accurate and reliable method for tracking effects of manure management on groundwater nitrate.

Acknowledgements

The authors thank the following people and organizations for their contribution to this study.

- The dairy producer who generously provided access to the grass field for groundwater, soil, manure, and crop sampling for more than 6 years. The producer also provided detailed information on his management practices during the study.
- Lynn VanWieringen, Washington State University, conducted soil, grass, manure, and irrigation sampling, coordination, and data analysis.
- The Washington State Department of Agriculture provided financial support for soil nitrate analyses.
- Chris Clark and George Boggs, Whatcom Conservation District, provided technical assistance.
- Nora Mena and Virginia Prest, Washington State Department of Agriculture, provided technical assistance.
- Deb Davidson, Washington State University, provided field assistance.
- Maury Mason and Seth Book, volunteers, provided field assistance.
- Washington State Department of Ecology staff:
  - Charles Piz provided technical advice, insights, and editing.
  - Manchester Environmental Laboratory: Nancy Rosenbower, Dean Momohara, and other analysts.
  - Field Assistants: Tanya Roberts, Sabrina Payne, Bruce Barbour, Mak Kaufman, Tom Buroker, Brian Pickering, Jodi England, Pam Marti, Maggie Bell-McKinnon, Becca Conklin, Bill Kammin, Kelli Slattery, and Kirk Sinclair.
  - Bellingham Field Office provided field and technical assistance: Mak Kaufman, Kelly Slattery, and Doug Allen.
  - Other reviewers and advisors: Martha Maggi, Kirk Sinclair, and Melanie Redding.
- Former Washington State Department of Ecology staff :
  - Denis Erickson, Darrel Anderson, Richard Grout, Devon Reid, Nicoleta Cristea, Ann Butler, Jamison Backous, Sara Livingston, Pam Covey, Andrew Craig, and Bernard Strong.

Executive Summary

Government, community, and agricultural groups around Washington State have been working to determine the main causes of nitrate contamination in groundwater and to identify cost-effective ways to address the problem. One of the dominant sources of nitrate loading to groundwater in the state, and around the United States, is nitrogen releases from agricultural practices.

The northwestern portion of Whatcom County is an area of high-intensity agricultural production. The main agricultural businesses in Whatcom County are dairy farming and berry production. Conventional practice for both types of operation is applying nitrogen-bearing fertilizer. Whatcom County has the second highest number of dairy cows in the state and the highest intensity of raspberry cultivation in the country.

Groundwater supply in this area is derived almost exclusively from the Sumas-Blaine Aquifer (SBA), an unconfined aquifer occurring in the unconsolidated glacial deposits that blanket the region (Figure ES-1). Over the last 30 years, this area has had one of the highest percentages of water supply wells in the state failing to meet the drinking water standard for nitrate (29% of wells tested had concentrations greater than 10 mg/L as nitrogen). Groundwater is the only source of drinking water for residents living in the northern, rural part of the county. As of 2010, the population living over the SBA not on city water systems was 18,000 to 27,000 people.

Factors that make groundwater in Whatcom County particularly sensitive to water quality impacts from intensive agricultural production include:

- Shallow depth to water.
- Relatively permeable character of the aquifer deposits.
- Long period of heavy rainfall each year.

Combined with the high mobility of nitrate in the environment, these characteristics enable rapid transport of nitrate from surface soils to the water table.

The Washington State Dairy Waste Nutrient Management Act[1] of 1998 requires that all dairies have approved Dairy Nutrient Management Plans (DNMPs) in place by 2003. The primary objective of the law is to ensure that surface water and groundwater quality in the state are not adversely affected by dairy manure.

The study described in this Executive Summary is one of a series of assessments the Washington State Department of Ecology (Ecology) has conducted over several decades to characterize the extent and nature of groundwater nitrate conditions in the SBA. We hope that insights gained from this study will guide stakeholders and decision-makers in efforts to restore and protect local and state groundwater resources.

[1] Chapter 90.64 Revised Code of Washington (RCW)

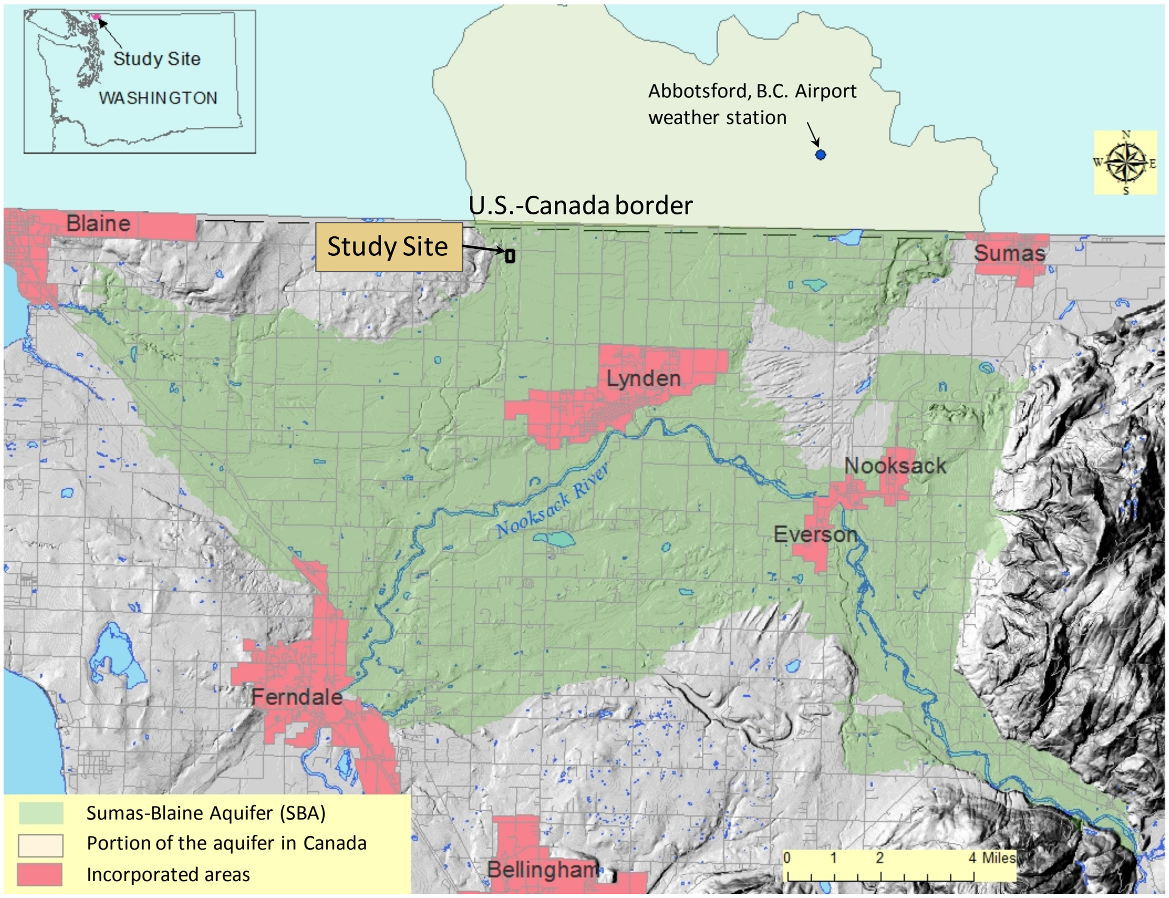

Figure ES-1. Sumas-Blaine Aquifer and study site location.

Purpose and objectives of the study

In 2003, Ecology's Bellingham Field Office requested that Ecology's Environmental Assessment Program conduct a field study to evaluate the effectiveness of DNMPs in protecting the quality of the SBA. One of the measures required in DNMPs is that producers collect samples of soil nitrate at the end of the growing season to assess nutrient balance. These soil samples were intended to provide a general assessment of nutrient balance at a field, not for assessing groundwater protection.

The main purposes of the study were to:

- Improve our understanding of the nitrogen dynamics and fate at a manured field.
- Evaluate the effectiveness of land application guidelines of DNMPs in protecting the quality of the SBA at the study site.

The technical objectives of the study were to:

- Track nitrogen dynamics and fate at a grass field overlying the SBA that receives manure applications:
    - Quantify the sources of nitrogen input and output to the study field on an annual basis.

- Quantify changes in soil nitrate and groundwater quality conditions at the study field over time.
- Compare the soil and groundwater results to Washington State guidelines and standards.

- Identify the key farm management and environmental factors that influence groundwater and soil nitrate conditions.
- Perform an annual nitrogen mass balance evaluation for the study field.
- Use the nitrogen mass balance and soil nitrate sampling results to estimate the annual nitrogen residual mass available for leaching to groundwater.
- Evaluate the correlation of the nitrogen residual estimates to groundwater results, and determine reliability of these estimates for predicting groundwater quality responses to manure management practices.
- Evaluate whether current guidelines for manure management and soil monitoring are adequately protective of groundwater.
- Evaluate study findings in the context of the SBA as a whole.

Background

The study site is a 22-acre grass field located in northwestern Whatcom County, Washington, about 3 miles north of the town of Lynden and 0.3 mile south of the Canada border (Figure ES-1). The site lies on the flat Lynden Terrace, a glacial outwash plain that slopes gradually southward to the Nooksack River.

The regional climate is humid maritime with mild temperatures and high wintertime precipitation due to proximity to the Pacific Ocean. Annual local precipitation ranges from 32 inches in the southwest part of the SBA to over 60 inches in Abbotsford, British Columbia, Canada. Roughly 60 to 70% of the annual precipitation typically occurs from October through March, outside of the typical growing season for crops.

Hydrogeology and soils

The SBA is the uppermost hydrogeologic unit in the area, covering about 150 square miles. The aquifer consists of unconsolidated sand and gravel outwash with minor clay lenses. The aquifer averages 50 feet in thickness. The depth to water is less than 10 feet over all but a small portion of the aquifer in the east, making the aquifer highly susceptible to surface contamination. The underlying hydrogeologic units are not feasible sources for large-scale drinking water consumption. Recharge to the SBA occurs mainly from precipitation that occurs from September through March.

The study site, located on the western edge of the SBA, is an area dominated by finer-grained material at the surface compared to other aquifer locations. The depth to the bottom of the aquifer is 40 feet at the site based on well borings from the site. Hale silt loam soil overlies the study site. Hale soils are part of the Lynden-Hale-Tromp grouping that overlies much of the SBA.

Dairy field management

The 22-acre study field has received manure for over 20 years at about the same application rate (~400 to 700 lb total N/acre/year) as during the study, according to the dairy producer. The site was planted in grass before the study and was tilled and re-seeded back to grass in April 2004 using conventional tillage practice. During the study, the dairy producer managed the field as before the study. Manure was mostly applied using an aerator (also referred to as subsurface deposition).

Manure was typically applied 3 to 5 times per year following each grass cutting. The final manure application for any given study year occurred between the end of August and early October. In 2 of the 4 study years (2005 and 2006), the final manure application occurred after the last crop harvest.

Each summer, irrigation water from a nearby shallow well was applied at the study site. The grass crop was harvested 4 to 5 times each year.

Study design

Ecology's Environmental Assessment Program partnered with the Washington State University Livestock Nutrient Management Program (WSU) to design an intensive, multi-media, multi-year monitoring study at a grass field overlying the SBA that received applications of manure. Participants from WSU focused on monitoring and characterizing the manure, soil, and crop, while Ecology focused on monitoring groundwater conditions underlying the study field.

We calculated the nitrogen mass balance of the study field each year and compared the estimated residual nitrogen to the shallow underlying groundwater nitrate concentrations. We also evaluated the effect of various environmental and management factors on the nitrogen mass balance and groundwater nitrate concentrations.

Our monitoring program focused on the following components of the nitrogen cycle to evaluate the balance of nitrogen at the study field system as shown in Figure ES-2:

- Inputs
    - Manure and inorganic fertilizer (mass of nitrogen applied to the field)
    - Irrigation water (volume and nitrogen concentration added to the field)
- Outputs
    - Grass harvested (mass of nitrogen removed from the field)
- Residual
    - Soil (fall nitrate mass in the top 1-foot of soil)
    - Groundwater near the top of the water table (nitrogen concentration)

To support the evaluation of nitrogen transport at the study site, additional field work was conducted to characterize the hydrogeology and soil characteristics of the study site. This included:

- Measuring static water levels in monitoring wells.
- Testing the hydraulic characteristics of the aquifer underlying the study field.
- Conducting grain size analysis of site deposits.
- Measuring chloride in groundwater to use as a conservative (non-reactive) tracer.
- Measuring other water quality constituents in groundwater that contribute to understanding nitrate levels.

An important factor that influenced the study results, although not part of the study design, is that the dairy producer tilled the grass field and replanted it in grass in April 2004, 4 months before the study began. Tillage typically increases nitrate mineralization and is often followed by increased nitrate leaching.

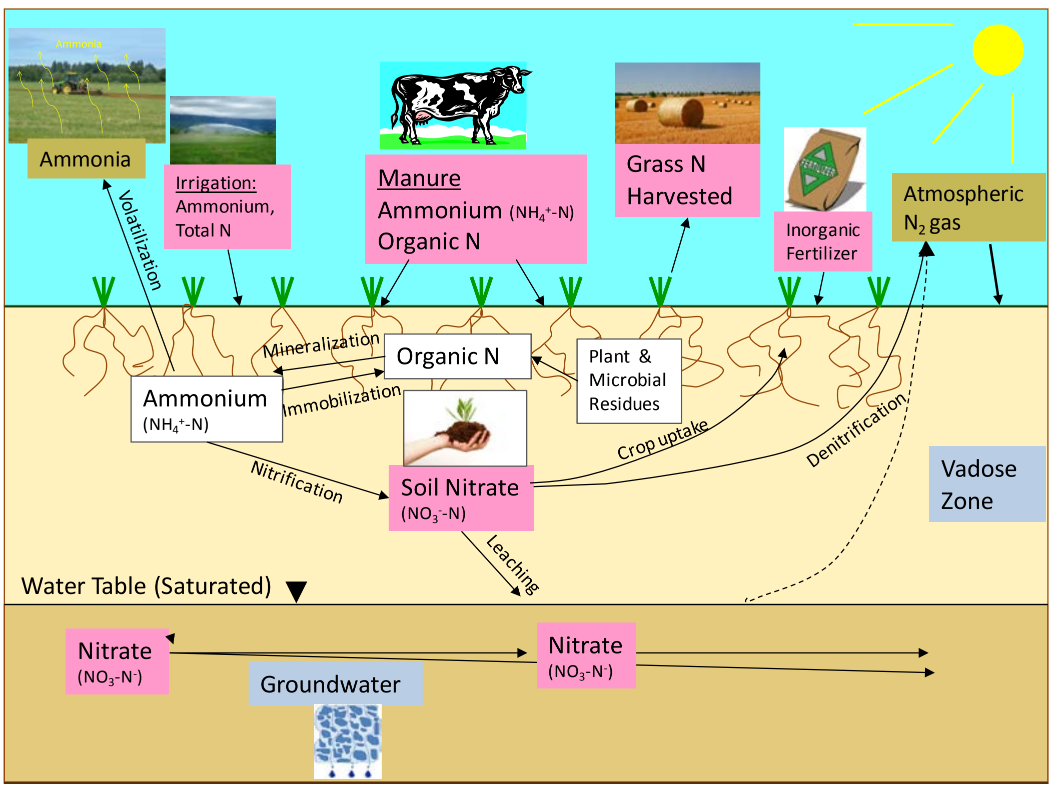

Figure ES-2. Major nitrogen components in the study field.

Media in pink boxes were monitored. Items in brown boxes were estimated.

Major findings

Groundwater sampling results

Groundwater sampled near the top of the water table represented recently recharged groundwater. Nitrate concentrations in individual monitoring wells varied widely on most dates (Figure ES-3). Fifty-six percent of the average monthly groundwater nitrate values over the course of the study were above 10 mg/L-N.

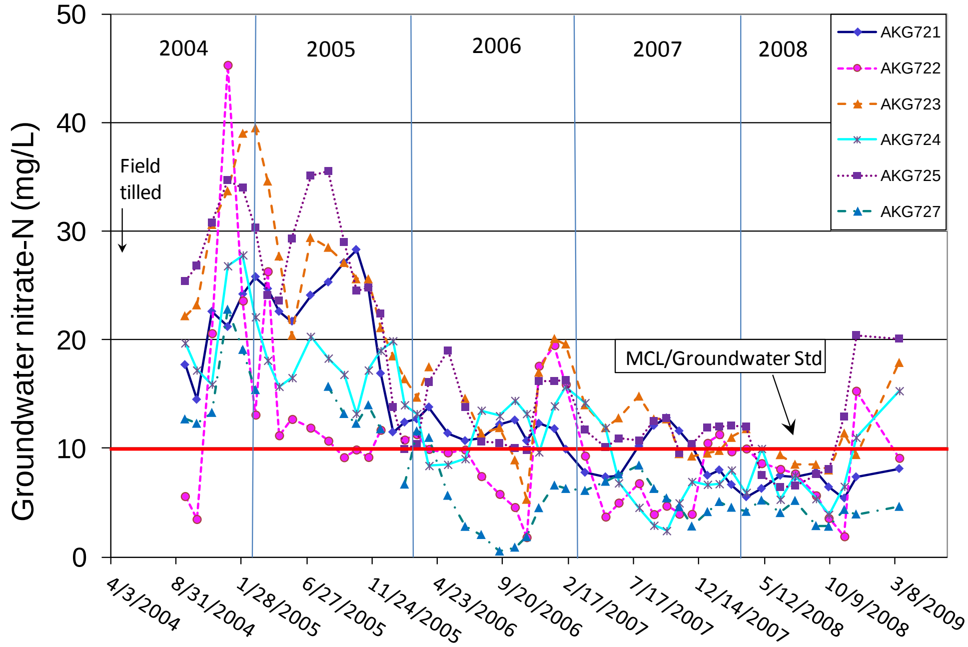

Figure ES-3. Shallow groundwater nitrate-N concentrations in individual monitoring wells.

Nitrate concentrations did not meet the groundwater quality/drinking water standard at the beginning of the study in 2004. The field had recently been tilled and replanted, which contributed to higher nitrate loading to groundwater than normal. However, ratios of groundwater nitrate-N (nitrate as nitrogen) to chloride indicated the main cause for the initial high concentrations was nitrogen loading from manure application. Groundwater nitrate gradually declined through the summer of 2008 due to lower manure application. At the end of the growing season in 2008, nitrate concentrations began to increase again, rising above the drinking water standard in 4 of the 6 monitoring wells.

Influencing factors

Several environmental and management factors influenced nitrate concentrations in soil and groundwater during the study.

Environmental factors

Hydrogeologic conditions

The depth to water at the site ranged from 0 to 11 feet below ground surface with the highest water table conditions occurring during the wet non-growing season (October through March). This allowed for rapid transport of available nitrate to groundwater.

Fine-grained material at the site likely influenced the substantial nitrogen loss observed in 4 of the 6 monitoring wells via microbially-mediated denitrification, which is less likely in coarse-grained materials.

Soil conditions

Higher moisture content in soils during the wet season generally led to higher permeability and rapid nitrate leaching. Summertime drying, which we observed in the beginning of the study, can slow mineralization and diminish crop uptake. This in turn can increase the amount of residual nitrate available to leach to groundwater in the fall. When irrigation water was applied earlier in the 2007 growing season, subsequent groundwater nitrate concentrations were lower than similar years without irrigation.

Heavy rainfall in the fall apparently led to a sharp increase in soil nitrogen mineralization at a time when crop growth was slowing. Most of the newly mineralized nitrate apparently leached to groundwater.

Precipitation

Most of the annual precipitation in the area occurred during a period of limited crop growth (October through March). Precipitation that infiltrated through the soil to the groundwater (recharge) carried available nitrate to the water table.

Upgradient conditions

Groundwater at the water table represents the most recently recharged water. We constructed all but 1 of the 7 monitoring wells with open intervals intersecting the water table. This allowed us to track groundwater responses to nitrate entering the aquifer in the near vicinity of the well. An analysis of groundwater travel times indicated that samples collected from the shallow monitoring wells represented recently recharged water that likely entered the aquifer within the study field boundaries. Based on this and other lines of evidence, we concluded that upgradient influences on groundwater quality results were probably limited.

Management factors

Rate of nitrogen application (External loading)

The more nitrogen applied to the field in excess of the crop demand, the higher the amount of nitrate that reached the water table in our study. Higher shallow groundwater nitrate concentrations occurred in the early winter following years with nitrogen application in excess of crop uptake (2005, 2008); concentrations were lower when the amount of nitrogen applied was less than the crop uptake (2006, 2007) (Figure ES-4).

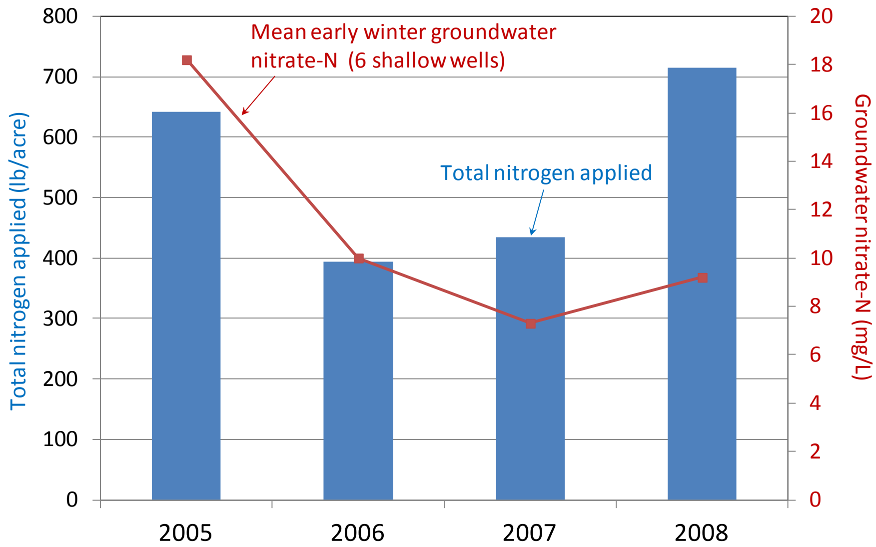

Figure ES-4. Total nitrogen mass applied annually (external loading) from manure and inorganic fertilizer (bars) and the mean early winter (November-December) groundwater nitrate-N concentration in 6 shallow wells (line).

Rate of nitrogen mineralization (Internal loading)

Organic nitrogen in the soil from previous years and manure organic nitrogen from the current year are bacterially converted each year to plant-available nitrogen. Accurately quantifying the amount and timing of this internal loading is difficult. The annual average estimated amount of mineralized nitrogen was roughly one-half of the annual mass of plant-available nitrogen (Figure ES-5).

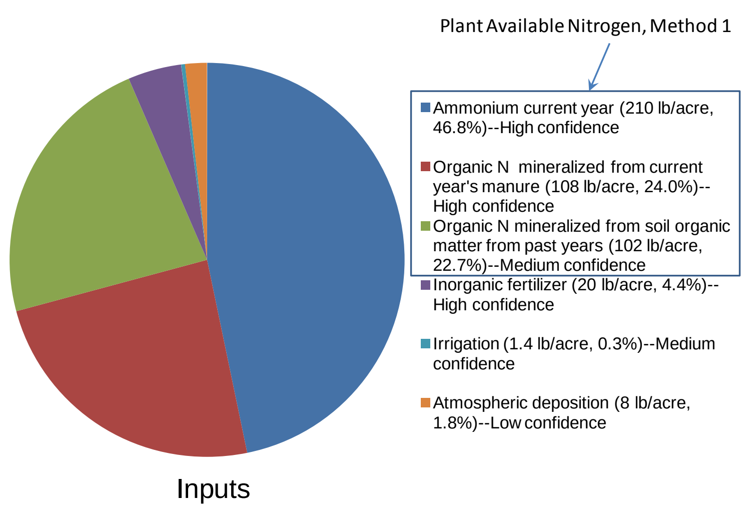

Figure ES-5. Average annual nitrogen inputs to the soil column during the study in lb/acre and our relative degree of confidence in the numbers.

Mineralization of organic nitrogen contributed to groundwater loading throughout the year, including the high-precipitation, non-growing season.

Tillage effects

Tillage of the study field in April 2004, 4 months before the start of the study, contributed to nitrate loading to groundwater the first year of the study but was not the dominant cause of high groundwater nitrate concentrations the first year.

Crop removal

Annual nitrogen removal in the crop was fairly consistent from year to year (393 to 457 lb/acre) despite large variations in the nitrogen application rate (394 to 715 lb/acre). This indicates that nitrogen was probably not the limiting factor for crop growth.

Timing of manure application

Applying manure too late in the growing season resulted in nitrate increases in groundwater. Similarly, applying manure too early in the season was followed by increased groundwater nitrate concentrations.

Denitrification

Denitrification in shallow groundwater at the site was controlled by reduced dissolved oxygen (DO) concentrations in 4 of the 6 wells. We estimated that an average of 28% of the nitrate in the 4 low-DO wells was converted to nitrogen gas and lost to the atmosphere. Denitrification was more prevalent in the summer and fall, when little oxygenated recharge water reached the water table.

Annual nitrogen residual estimates – correlation with groundwater nitrate

An important goal of the DNMPs is to minimize the amount of residual nitrogen left in the field after the growing season. This minimizes the amount of nitrate available to leach to groundwater. We evaluated 2 field-based methods of estimating the amount of residual nitrogen left in the study field at the end of each growing season:

1. Mass balance estimate of annual nitrogen residual
2. Fall soil nitrate residual (1 foot depth)

The main input variable for the mass balance analysis is the amount of plant-available nitrogen (PAN). We used the detailed nitrogen application data collected during the study with the most current method for estimating PAN for western Washington and Oregon.

Using the mass balance analysis, the annual nitrogen residual (N_{Excess}) from 2005 to 2008 was -79 to 146 lb/acre with a 4-year average of 19 lb/acre (Figure ES-6). Significant nitrogen deficits were predicted for 2006 and 2007. The annual mass balance residual did not correspond well with the nitrogen application rates or with groundwater concentrations. The main source of uncertainty in the mass balance evaluation was the contribution from mineralized organic matter from past years.

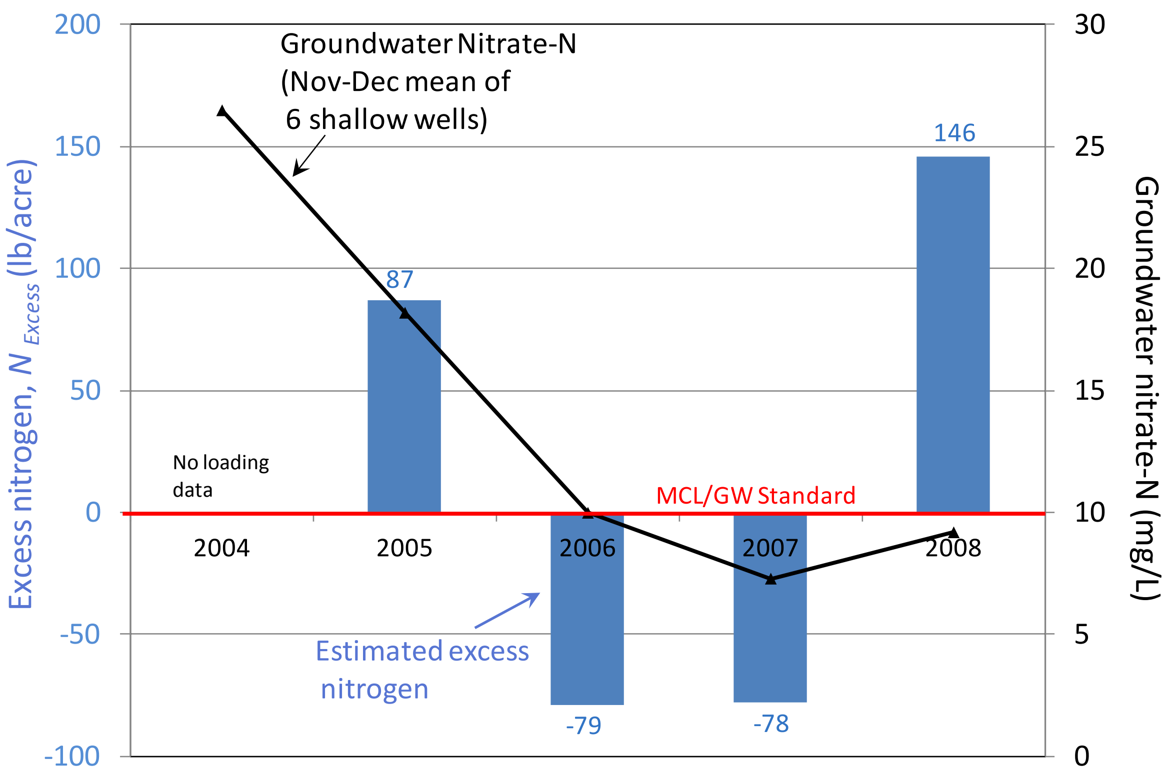

Figure ES-6. Estimated excess nitrogen N_{Excess} (bars) calculated using mass balance vs. annual mean winter (November-December) groundwater nitrate-N concentrations in 6 shallow wells (line).

r^2=0.22
MCL = Maximum contamination level
GW = Groundwater

The second method used the fall soil nitrate values collected weekly in September and October to estimate the annual mean and maximum soil nitrate residual. Soil nitrate concentrations varied greatly over time as shown in Figure ES-7. The 4-year average fall (September-October) soil nitrate result was 72 lb/acre; the maximum was 104 lb/acre. These averages were 4 to 5 times the mass balance average and were highly variable over short time periods. Soil nitrate residual values did not track closely with nitrogen application rates.

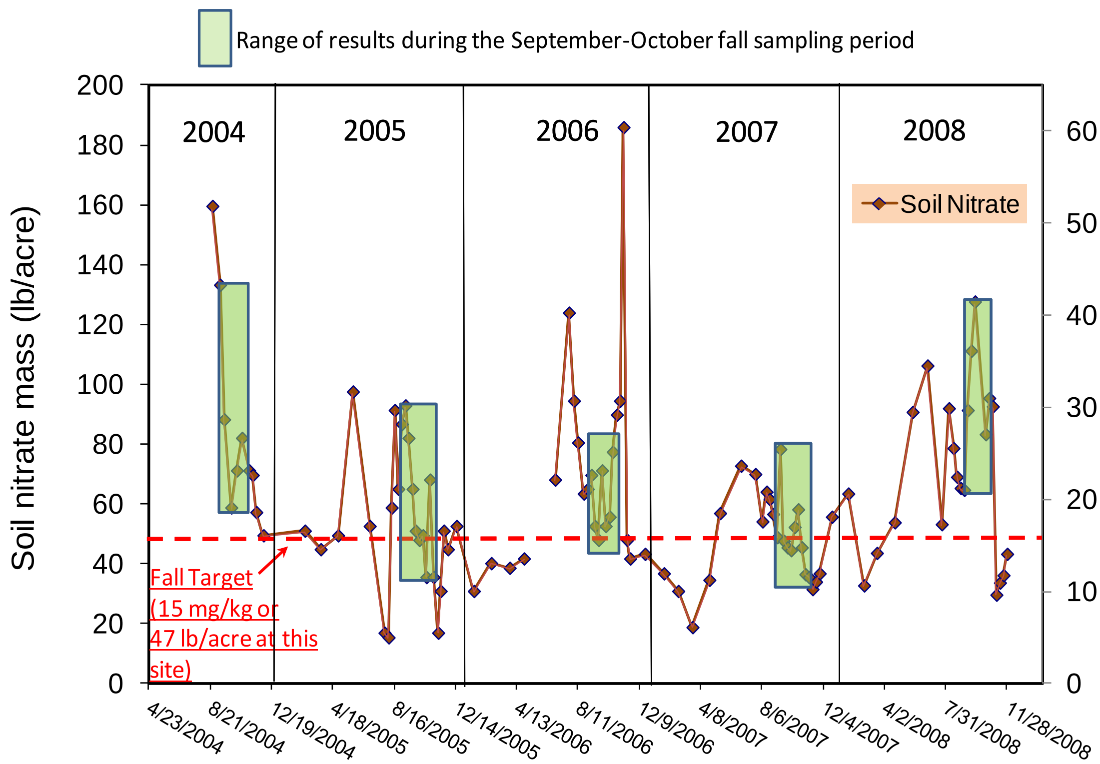

Figure ES-7. Soil nitrate results at 1-foot depth.

Green shaded areas indicate results for the typical fall soil sampling time, September through October. Red dashed line indicates the level below which management changes are not recommended based on Sullivan and Cogger (2003).

Fall soil nitrate results are typically an underestimate of the amount of nitrate available for leaching at the end of the growing season, because any nitrate already leached cannot be measured. In addition, nitrate deeper in the soil, as well as nitrate generated in the soil after October 31, are not included in fall soil nitrate results.

We attempted to correlate results of the nitrogen mass balance method and the fall soil nitrate method with early winter groundwater nitrate concentrations, because early winter is typically the time when the end-of-season residual nitrate in the soil reaches the water table. Neither approach correlated well with groundwater nitrate concentrations, as shown in Figures ES-6 and ES-8.

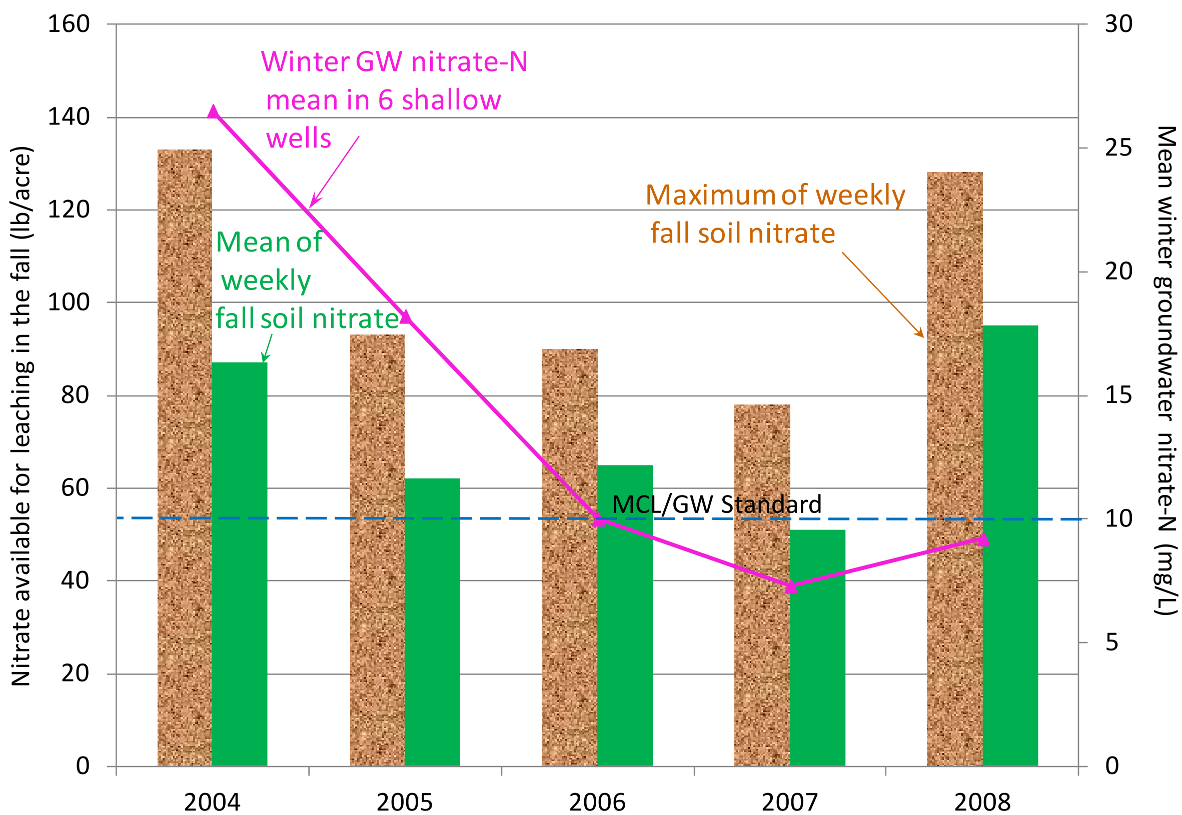

Figure ES-8. Comparison of fall soil nitrate mean concentrations (green untextured bars) vs. maximum concentrations (brown textured bars) with mean winter (November-December) groundwater nitrate-N concentrations in 6 shallow wells (line).

r^2=0.30 for maximum fall soil nitrate.
r^2=0.07 for mean fall soil nitrate.

Because mass balance and soil nitrate methods were not good predictors of nitrate leaching, we used an alternative method for estimating residual nitrate. The GWNO3-BACKCAST model was used to predict the amount of nitrate necessary to produce the groundwater nitrate concentrations observed. The model is based mainly on measurements at the study site (recharge rates, groundwater nitrate concentrations, hydraulic conductivity, and horizontal gradients).

On average, the BACKCAST model back-calculated an annual nitrate loading to groundwater of 115 lb/acre during the wet season, with a range of 42 to 230 lb/acre (Figure ES-9). Seasonal back-calculated averages were 66 lb/acre during the late fall/early winter and 49 lb/acre during the late winter/early spring. The BACKCAST model predictions were most sensitive to the rate of recharge, the denitrification rate in groundwater, and the thickness of the groundwater mixing zone.

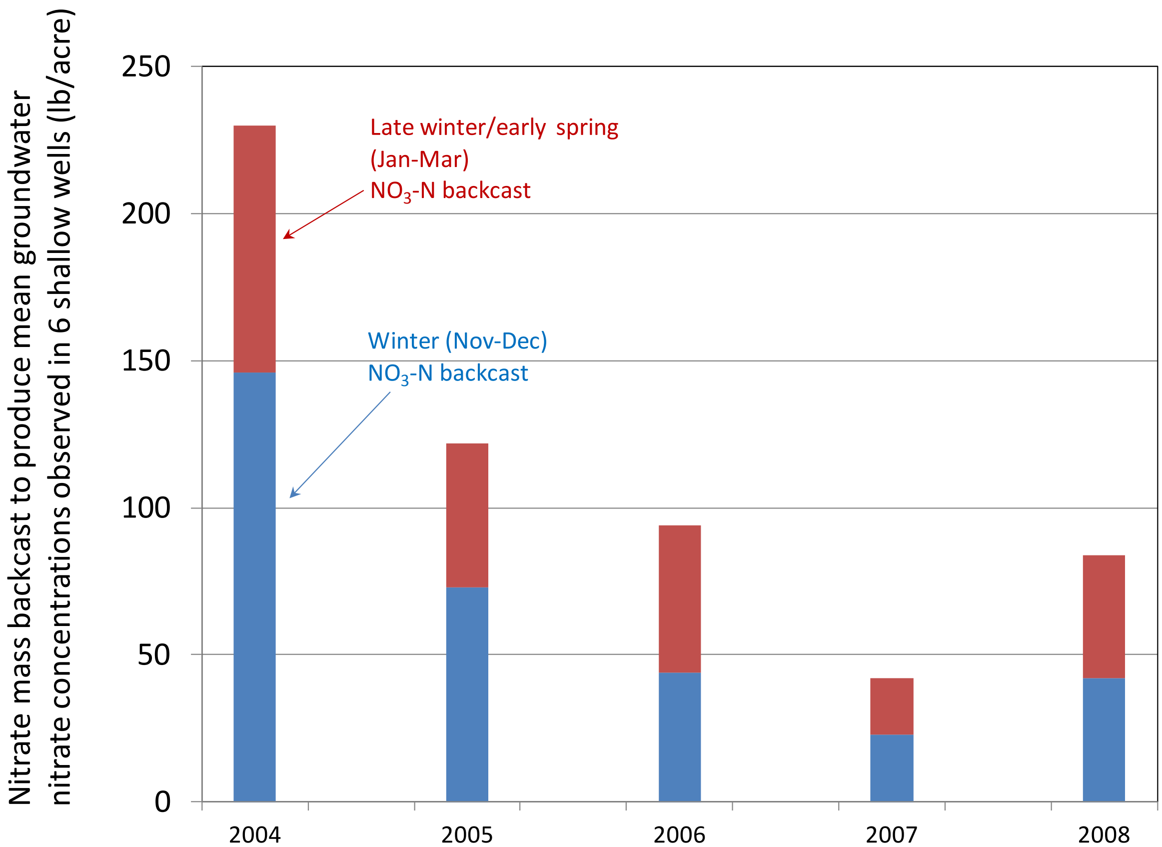

Figure ES-9. Total wet-season nitrate mass load predicted using the GWNO3-BACKCAST model.

Soil nitrate as an indicator of leaching to groundwater

We evaluated the target post-harvest soil nitrate concentration recommended for dairies in western Washington, 15 mg/kg, in relation to the recharge amounts observed. The calculated leachate nitrate concentration that would result from combining the fall soil nitrate threshold concentration for grass (15 mg/kg) with the observed annual volumes of recharge ranged from 11 to 18 mg/L-N. This does not include nitrate available for leaching below the top foot. All soil nitrate samples collected according to the protocols for post-harvest soil nitrate testing during the study exceeded the 15 mg/kg target.

If the seasonal *average* September to mid-November soil nitrate concentrations was used in the same calculations with the recharge that occurred during the study, the estimated nitrate concentration in leachate would have been 16 to 23 mg/L-N. If the seasonal *maximum* September to mid-November soil nitrate concentration was used in the calculation, the estimated leachate nitrate concentration would have been 29 to 50 mg/L-N.

Recommendations

Reducing nitrate leaching to groundwater at manured dairy fields, like the one in this study over the Sumas-Blaine Aquifer (SBA), will require improving manure application based on evolving science and technology. This includes nitrogen loading analyses that take groundwater into account. Groundwater monitoring will be needed to evaluate the effectiveness of measures to reduce nitrate loading to groundwater.

Based on the results of this study, the following actions are recommended to promote improvements to groundwater quality in the SBA and in other areas of Washington State with similar conditions.

Reduce nitrate loading to groundwater

- Develop a process whereby (1) manure and fertilizer nitrogen inputs and outputs are tracked on a field-by-field basis and (2) the information is used to minimize nitrate leaching below the root zone. Because of high seasonal rainfall and shallow depth to water in the area, appropriate timing and amount of nutrient application is crucial. Involvement of state and local organizations in partnership with universities, dairy and other agricultural producers is needed to improve nitrogen use efficiency and protect groundwater quality.

- Review available mass balance assessment methods for use in Dairy Nutrient Management Plans (DNMPs). Develop or adapt existing methods so that they more accurately account for effects on groundwater (i.e., more accurate assessment of soil organic matter contribution).

- Calculate nitrogen applications (manure, fertilizer, irrigation water) and removal (crop removal) based on measured amounts and nitrogen analyses to estimate mass balance. This is especially important in areas where groundwater nitrate already does not meet the drinking water standard. Timing of nitrogen application relative to recharge is especially important.

- Our results indicate that it is best to schedule the last manure application by late August to early September. Manure application during the high-recharge period (September through mid-March) is likely to increase nitrate leaching to groundwater.
- Where groundwater is well-oxygenated and denitrification rates are low, take special care to prevent overapplication or application during the high-recharge period.
- If soil moisture is low in the summer, consider irrigating to increase mineralization and nitrification which may increase available nitrate for the crop. Avoid overapplication of irrigation water to prevent nitrate leaching.
- Consider extending the time between tillage events to decrease the amount of nitrogen reaching groundwater.
- Consider limiting manure application to forage crops during the first season following tillage.
- Consider updating the post-harvest soil nitrate test (PSNT) guidance to incorporate hydrologic influences (i.e., local aquifer recharge) based on the expertise of land grant universities as well as local and state scientists.
- Until revised, use the existing post-harvest soil nitrate protocols (methods and timing) and targets (15 mg/kg for grass; 20 mg/kg for corn) as criteria for evaluating manure management at dairy operations in western Washington.

Monitor to evaluate the effectiveness of management improvements

A program is needed to determine how well current and future manure management practices are working to improve groundwater quality. Because there is no reliable substitute, direct groundwater monitoring using dedicated monitoring wells is a key component of an effectiveness monitoring program.

Although groundwater monitoring is the only available way to determine the amount, or the concentration of, nitrate that actually reaches the water table, fall soil nitrate monitoring is a necessary tool for on-farm nutrient management. If conducted with limitations in mind, soil nitrate monitoring also can serve as a screening tool to focus closer inspection of groundwater conditions.

Nitrogen mass balance evaluations also are an important tool for dairy producers to manage nutrients and identify potential courses of action to address high soil and groundwater nitrate concentrations. Current methods for analyzing mass balances for DNMPs should be evaluated to ensure that the methods are as accurate as possible.

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Introduction

Background information

The Sumas-Blaine Aquifer (SBA) underlies about 150 square miles of U.S. land and is the primary source of drinking water for 18,000 to 27,000 residents of northwest Whatcom County, Washington (U.S. Census, 2010). The SBA is part of the larger Abbotsford-Sumas Aquifer that straddles the U.S.-Canada border. The aquifer averages 50 feet in thickness in the U.S. (Figure 1; Tooley and Erickson, 1996).

Groundwater within the SBA flows predominantly from north to south (British Columbia, Canada to the U.S.), but local patterns of flow are also affected by interactions with surface water features (Figure 1). The depth to water is less than 10 feet in most of the SBA but is more variable in British Columbia. In winter the depth to water is near the surface in much of the SBA, requiring artificial drainage to prevent flooding due to heavy precipitation in some places.

Nitrate in drinking water

Nitrate concentrations in excess of 10 mg/L-N, the maximum contaminant level (MCL) acceptable in public drinking water supplies[2] and the groundwater quality standards for Washington,[3] have been documented in the SBA for at least the past 23 years (Erickson and Norton, 1990; Garland and Erickson, 1994; Cox and Kahle, 1999; Erickson, 2000 and 1998; Carey, 2002; Almasri and Kaluarachchi, 2004; Mitchell et al., 2005; and Redding, 2008).

In 1997, 21% of 250 private wells tested in the SBA exceeded the drinking water limit (Erickson, 1998). In a 35-well subset of the 250 wells previously sampled, 71% contained nitrate at concentrations higher than 10 mg/L-N between 2003 to 2005 (Redding, 2008).

Several public water supply wells near the City of Lynden exceed the drinking water standard for nitrate, affecting over 1,000 residents (Pell, 2011).

Agricultural activities in the U.S. and British Columbia, Canada

Intensive agriculture has been conducted over the SBA for the past 50 years. Dairy farming has historically been the predominant agricultural activity over the aquifer, with raspberry and other berry production becoming more prominent in the past 20 years. There are approximately 37,000 acres in dairy production, 8,200 acres of raspberries, and 2,600 acres of blueberries in Whatcom County (Embertson, 2010; Whatcom Farm Friends, 2012).

Berry and poultry production have replaced most of the dairy land in the Abbotsford area of British Columbia. Zebarth et al. (1998) showed that much of the surplus nitrogen that leaches to groundwater or runs off to surface water on the Canadian side of the aquifer is due to changes in

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[2] Chapter 246-290-310 Washington Administrative Code (WAC)
[3] Chapter 173-200 WAC

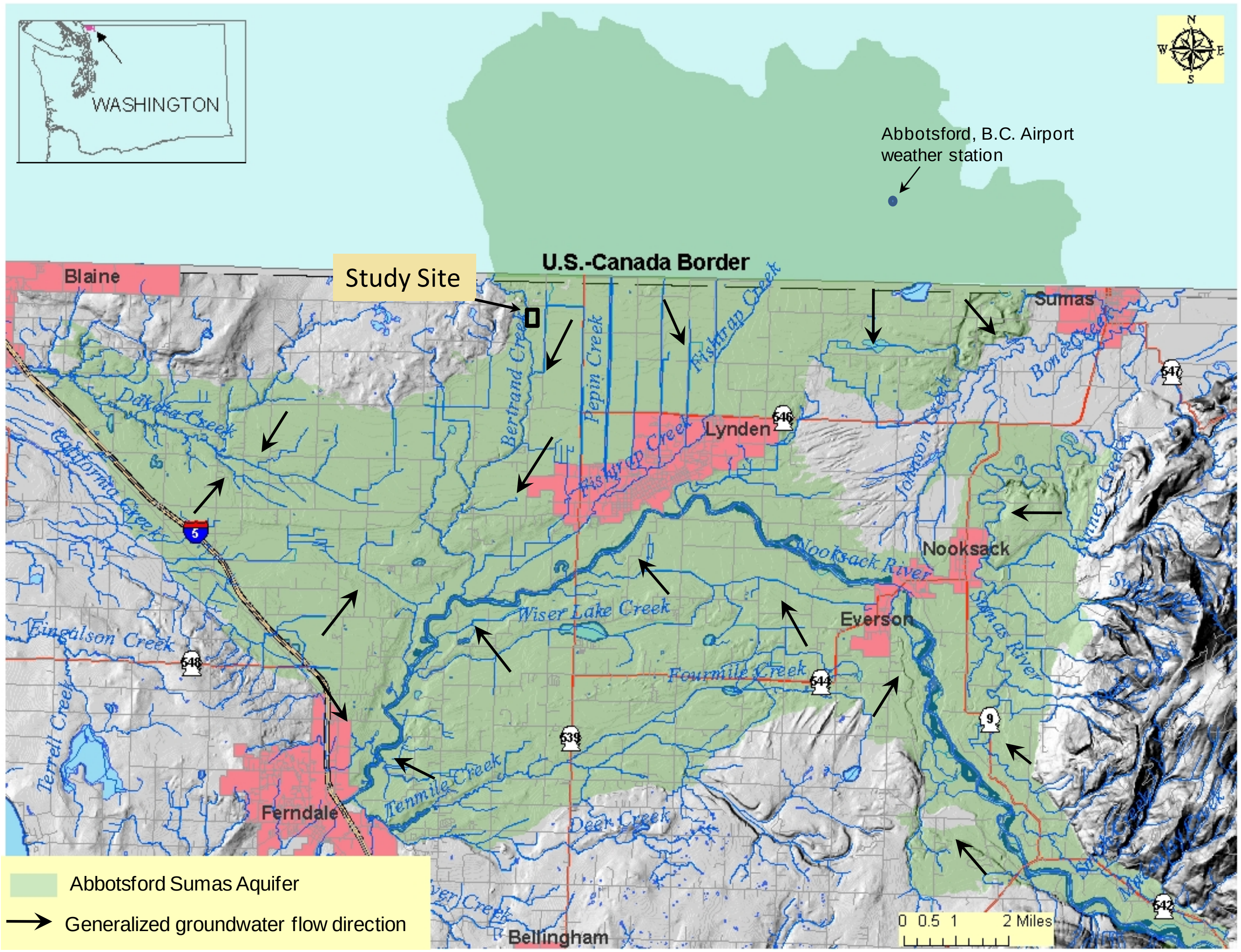

Figure 1. Study site location within the Abbotsford-Sumas Surficial Aquifer.
Groundwater flow direction arrows are from Erickson (1998) and Graham (2008).

agricultural practices over the past 40 years. Small fruit crops, which have replaced almost all of the cropland formerly in grass for dairy cows, take up only 10% of the amount of nitrogen taken up by forage crops, leaving more nitrogen available for infiltration below the root zone (Zebarth et al., 1998). Currently there are about 2,500 acres in raspberries and 1,000 acres in blueberries over the Canadian portion of the aquifer (Sweeney, 2012).

On the Washington side of the aquifer, the number of dairy farms dropped by one-half from roughly 2000 to 2010 (Embertson, 2010). Fields formerly planted in grass to feed dairy cows are being converted to crops that take up less nitrogen and, as a result, contribute a surplus of nitrogen to groundwater similar to that on the Canadian side of the aquifer.

Agricultural activities overlying the Canadian portion of the aquifer have also resulted in groundwater quality impacts (McArthur and Allen, 2005). The concentrations of nitrate along the Canada-U.S. border area are variable, with the highest concentrations on the eastern side of the aquifer.

Although the distribution of nitrate concentrations in groundwater entering the U.S. from the Canadian side of the aquifer system remains uncharacterized, groundwater typically flows horizontally in the direction of flow (generally north to south in the SBA) with solute concentrations dispersing deeper into the aquifer with distance from the source. Therefore, shallow groundwater in the U.S. would most likely not be affected by activities north of the border.

Adverse effects of high nitrate concentrations

High nitrate concentrations in drinking water can cause methemoglobinemia, or blue-baby syndrome, in infants. This potentially life-threatening condition is caused by conversion of nitrate to nitrite in the digestive system. The nitrite then reacts with iron in hemoglobin, restricting transport of oxygen to cells. An increased risk of spontaneous abortion or certain birth defects may be associated with drinking nitrate-contaminated water. Cancer risks also have been associated with elevated nitrate in water and food (Centers for Disease Control and Prevention, 1996; Chiu and Tsai, 2007; Ward et al., 2005; Weyer et al., 2001).

Besides human health effects of nitrate, nitrate in groundwater can adversely affect surface water by increasing primary productivity in streams, rivers, and lakes hydraulically connected to the aquifer system. When algal and plant material that depend on nitrogen decompose, oxygen depletion can adversely affect fish and other aquatic life (Matson et al., 1997; Howarth and Marino, 2006).

Application of manure to crops

Much effort has gone into developing nutrient management plans for dairies in the area, since the Washington Dairy Nutrient Management Act was adopted in 1998[4]. Yet questions remain about the best management practices necessary to simultaneously maintain crop health and reduce and prevent nitrate contamination in local groundwater. Because the SBA already displays a high vulnerability to nitrate leaching (Erwin and Tesoriero, 1997), and so much of the land overlying the aquifer receives dairy nutrients in the form of liquid manure, it is important to optimize nutrient management. Some of the issues of concern for land application of manure include:

- Rate of nitrogen application
- Timing of manure application
- Soil type (texture and organic matter influence)
- Methods for estimating plant-available nitrogen
- Methods for evaluating excess/deficit nitrogen (soil nitrate, leachate nitrate concentration, groundwater nitrate concentration)

While one goal of manure application is to apply an amount of nitrogen that will contribute to optimal crop growth, simultaneously achieving a close balance between inputs and outputs of nitrogen to protect groundwater quality is often elusive.

A number of studies have shown that measured concentrations of nitrate in soil or soil pore-water, or estimates of surplus nitrogen loading from mass balance surveys, are not reliable predictors of underlying groundwater nitrate concentrations (Viers et al., 2012; van der Schans et al., 2009; van Es et al., 2006; Basso et al., 2005; Zebarth et al., 1998; Bechmann et al., 1998). These methods can either overestimate or underestimate groundwater impacts. This is because transformations between various forms of inorganic and organic nitrogen are difficult to predict. However, Goss and Goorahoo (1995) found that although farm nitrogen budgets did not accurately predict groundwater nitrate concentrations, they were useful for identifying farms likely to cause environmental contamination.

The timing and amount of manure applied to crops have been found to be the key factors in maintaining nitrogen balance on manured fields (Oenema et al., 2010; Van Es et al., 2006; Verloop et al., 2006; Di and Cameron, 2002).

The amount and timing of precipitation that carries soil nitrate to the water table also plays a significant role in the concentration of nitrate ultimately reaching the water table (Sonneveld et al., 2010; Oenema et al., 2010; de Ruijter et al., 2007; Boumans et al., 2005; Zebarth, 1998). Smith et al. (2002) suggest that liquid manure applications not be made during wet winter months in “Nitrate Vulnerable Zones” of the United Kingdom, because heavy rainfall in winter months leaches soluble nitrate below the root zone.

[4] Chapter 90.64 Revised Code of Washington (RCW)

Nitrogen cycle

Nitrate is part of the dynamic system of nitrogen-containing compounds transformed in the environment and referred to as the *nitrogen cycle*. This section describes the major parts of the nitrogen cycle that occur at a typical manured field over the SBA.

Figure 2 shows the main components of the nitrogen cycle for the study site during the 2 main agricultural seasons.

- The top diagram in Figure 2 represents the spring/summer period, when the water table is several feet below the root zone of the grass crop. The vadose zone includes the root zone [roughly 0 to 3 feet below ground surface (BGS)] and extends to the water table, which during this drier season is roughly 10 feet BGS.
- The bottom diagram in Figure 2 shows the fall/winter scenario after heavy precipitation has raised the water table to within roughly 1 to 3 feet of the surface, frequently intersecting the root zone.

The first manure application to grass crops typically occurs in early spring, and additional applications are made after each cutting through the summer. The final yearly manure application usually occurs in September or October. The nitrogen composition of dairy manure varies depending on a number of variables including animal genetics, feeding programs, and available feed (ASAE, 2005).

Volatilization

A portion of the ammonium contained in manure converts to ammonia gas after application and volatilizes to the atmosphere. The amount that volatilizes depends on the application method, weather conditions (especially wind, rainfall, and temperature), and soil conditions. Most volatilization occurs during the drying process soon after manure is applied (Beegle et al., 2008; Sullivan, 2008). Therefore, rainfall or saturated soil conditions during or shortly after application can significantly limit volatilization. If ammonium infiltrates into the soil before drying, then less ammonia volatilizes than if drying had occurred. Chemical conditions in the soil also influence ammonia volatilization. Volatilization is higher where soil pH is high and cation exchange is low (Beegle et al., 2008).

Subsurface deposition, the principal method of manure application used at the study site, reduces the amount of ammonia volatilized to about 15% compared to 30% to 45% for spray application methods (Sullivan, 2008). Even with aerial spraying, ammonia volatilization decreases if manure is incorporated into the soil soon after application. Reduced ammonia volatilization using this method allows more of the nitrogen applied to be available for plant uptake or leaching to groundwater.

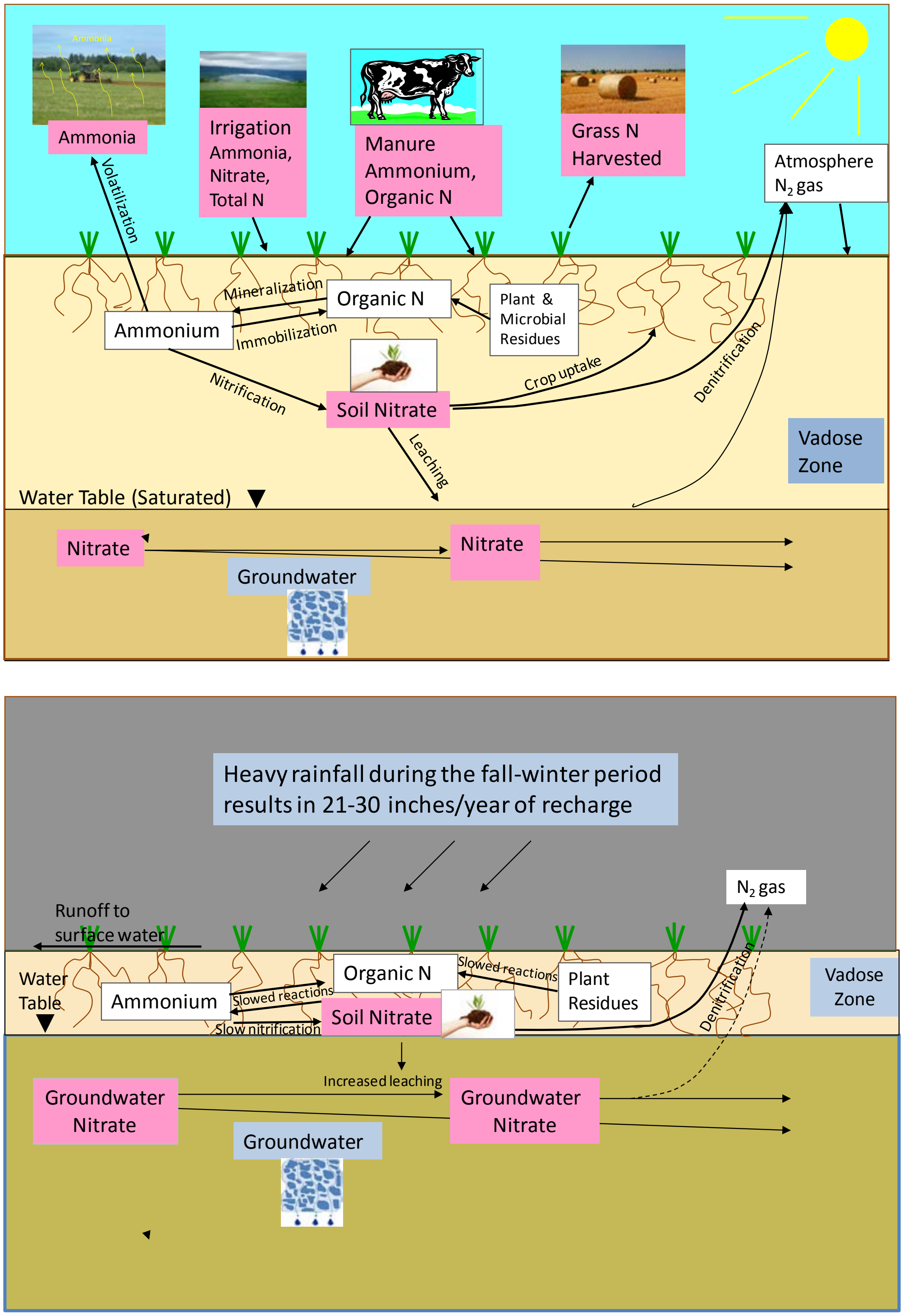

Figure 2. Major nitrogen transformations in spring/summer (top) and fall/winter (bottom) at the study site.

Media in the pink boxes were measured during the study.
Recharge estimate is from Cox and Kahle (1999).

Mineralization

Mineralization is the general term for conversion of organic nitrogen (non-plant available) to inorganic nitrogen (plant-available) by bacteria. The rate of mineralization is affected by temperature, soil moisture, and the redox condition of the soil. Roughly one-third to one-half of the organic nitrogen in land-applied manure mineralizes quickly to ammonium, while the more resistant portion converts gradually over time in a decay process (Beegle et al., 2008). This varies widely among manure types.

Mineralization is slower during the winter than during the warmer seasons (Trindade et al., 2001; Zhao et al., 2010; Cookson et al., 2002). The reverse process of mineralization, called *immobilization*, is the result of the uptake of ammonium or nitrate by microorganisms, temporarily making the nitrogen unavailable to crops. When the microbes die, immobilized nitrogen once again becomes plant available through mineralization.

Nitrification

Nitrification is the bacterial conversion of ammonium to nitrate. This is usually a rapid process favored by warm temperatures, adequate moisture, and aerobic conditions. Nitrification occurs throughout the year. The optimum temperature for nitrification in cultured bacteria from soil is in the range of 25 to 30º C; however, studies have shown that nitrification also occurs at colder temperatures typical of winter conditions (Norton, 2008).

Nitrification is limited in very wet and very dry conditions. In the summer, if the soil moisture becomes too low, bacteria become dehydrated and nitrification is severely slowed (Norton, 2008). Saturated winter conditions can also inhibit nitrification due to reduced oxygen.

Although optimal pH for mineralization is considered to be neutral to slightly alkaline, mineralization has been observed in soils with pH as low as 3.0 (Norton, 2008). Recent evidence suggests that mineralization occurs readily in acidic soils where blueberries are grown (Zebarth, 2013). The nitrification rate for Hale silt loam, the predominant soil type at the study site (pH 5.1-6.5), should not be limited by pH.

Crop uptake

Nitrogen is the main limiting nutrient for most crops. Nitrate is the most available form of nitrogen for plant root uptake (Olson and Kurtz, 1982). Positively charged ammonium ions react with negatively charged soil particles (particularly clay particles), keeping ammonium relatively stationary in soil. However, plant roots readily take up ammonium, if available, especially in the spring before nitrification increases. During the winter months, grass crop uptake rates are significantly slower than during the growing season (Hermanson et al., 2000).

Denitrification

Denitrification is the reduction of nitrate under anaerobic (negligible oxygen) conditions by bacteria to nitrogen gas. In soil, anaerobic conditions often occur under saturated conditions, due in part to the reduced availability of oxygen. Denitrification occurs when the rate of oxygen consumption, usually by bacteria, exceeds the rate of oxygen diffusion in the soil.

Denitrification requires the transfer of electrons from a donor such as organic carbon. Dissolved organic carbon is a component of organic material (including manure) and is the common electron donor for the reaction (Green et al., 2008; Desimone and Howes, 1996).

Organic carbon from manure can build up in the soil over time and enhance the denitrification potential in the soil (Hermanson et al., 2000). Manure itself commonly enhances anaerobic conditions by supplying highly labile forms of carbon that stimulate microbial activity and oxygen depletion (Zebarth, 2013). Nitrate can also be reduced either bacterially or chemically where iron or sulfur are the electron donors (Buss et al., 2005).

Like the nitrification process, the reaction rate for denitrification increases with temperature with an optimum in the range of 25 to 35º C (Buss et al., 2005). Rates of denitrification are known to be highly variable over small distances. *Hot spots* are often reported in soils where denitrification rates are much higher than rates in nearby locations (Coyne, 2008).

Denitrification can occur in both the vadose zone and in groundwater. Denitrification in groundwater is most likely when dissolved oxygen and oxidation-reduction potential are low and organic carbon is available (Singleton et al., 2007; Gillham and Cherry, 1978).

Leaching

In the fall and winter, percolating water due to heavy rain in western Washington transports residual nitrate in the soil and carries it past the root zone, through the vadose zone to the water table. Most, if not all, excess nitrate remaining in the soil after the growing season leaches to groundwater in the fall to early winter (October to January) (Beegle et al., 2008; Downing, 2008; Hermanson et al., 2000; Zebarth et al., 1998; Paul and Zebarth, 1997; Kowalenko, 1989 and 1987). Ammonium is either held in the soil or converted to nitrate and is therefore not normally found in the dissolved phase at the water table.

Leaching also occurs in the late winter/early spring, when precipitation exceeds evapotranspiration and plant uptake of nitrogen is low (Chesnaux et al., 2007; Zebarth and Paul, 1997). Trindade et al. (2001) found high nitrogen mineralization rates during the winter when soil temperatures were above 5°C and soil moisture was near field capacity. If not taken up by plants, this newly generated nitrate can be readily transported to the water table. Several recent studies indicate that winter nitrogen processes are more important than previously thought (Zebarth, 2013).

Leaching during the summer due to irrigation and preferential flow was not addressed in this study but could be a significant factor in the annual nitrogen cycle at the field (Nimmo, 2013).

Dairy nutrient management plans

The Washington State Dairy Waste Nutrient Management Act[5] of 1998 requires that all dairies develop DNMPs. The primary objective of the law was to ensure that surface water and groundwater quality in the state are not adversely affected by dairy manure. DNMPs were required to be approved by July 1, 2002, and implemented with final certification by December 31, 2003. These plans were submitted to local conservation districts for review and approval.

The Washington State Department of Agriculture is responsible for overseeing dairies and DNMPs. The minimum elements of a DNMP are described in the Department of Agriculture web site: http://agr.wa.gov/FoodAnimal/Livestock-Nutrient/DairyNutrientMgmtPlans.aspx

A primary goal of the DNMPs is to balance nutrient application and plant uptake on each individual farm. One aspect of this goal is that the amount of nitrogen removed in the crop match as closely as possible the amount of nitrogen available from the combination of manure nitrogen and nitrogen released from organic material in the soil.

DNMPs are required to outline steps necessary to ensure proper handling and use of dairy manure. Because most of the land on a dairy farm is manured fields, the focus on nutrient management is vital for addressing groundwater nitrate issues. This is particularly true in areas of known vulnerability to groundwater nitrate contamination, such as the Sumas-Blaine Aquifer.

DNMPs require that one composite fall-season soil nitrate analysis be taken at each field receiving dairy nutrients soon after the last harvest and before significant precipitation. Results of the fall (post-harvest) soil nitrate test are used to evaluate the balance between the amount of nitrogen available to the plant (from nutrient application as well as from the soil) and the amount removed by the crop (Sullivan and Cogger, 2003). Sullivan and Cogger established that changes need not be made to current management practices of manured fields if post-harvest soil nitrate is below 15 mg/kg (55 lb/acre typically).

The post-harvest soil nitrate target (15 mg/kg) was based on results of Washington State University studies in and near Puyallup, Washington (Sullivan, 2013) and was not designed to address impacts of nitrate leaching to groundwater. These studies indicated that maximum yields could be attained with 10 mg/kg fall soil nitrate. However, because median post-harvest soil nitrate results for grass fields in Whatcom County were in the 20 to 25 mg/kg range, a compromise of 15 mg/kg was chosen for the target.

The timing of fall soil nitrate sampling can have a critical effect on nutrient balance evaluations. For fine-grained soils in Whatcom County, such as Hale soils at the study site, the recommended time for post-harvest soil nitrate testing is after the last harvest and before 5 inches of precipitation has fallen after September 1. Results typically represent only what is left after at least a portion of the residual nitrate has leached below the sample depth (one foot). If sampling occurs before the last manure application for the year, the result would not include a potentially significant amount of nitrate that will be available for leaching during the rainy season.

[5] Chapter 90.64 RCW

Hirsch (2007) found that soil nitrate sampling after harvest did not capture all of the nitrate leached below the root zone. She recommended testing soil nitrate at the same time as harvest to avoid missing leaching losses. The potential for wide variability in soil nitrate results over short time spans suggests that the standard practice of collecting a single fall soil nitrate sample is a potentially poor predictor of the amount of nitrate that will ultimately reach the underlying water table.

Study Purpose and Objectives

The Bellingham Field Office of Ecology's Water Quality Program requested that Ecology's Environmental Assessment Program design and conduct a long-term study at a dairy farm to:

- Improve our understanding of nitrogen dynamics and fate at a manured field.
- Evaluate the effectiveness of the land application guidelines of Dairy Nutrient Management Plans (DNMPs) in protecting the quality of the Sumas-Blaine Aquifer (SBA) at one location.

The long-term study was designed to measure and document the sequence of changes in nitrogen at a grass field receiving manure in terms of soil, grass crop, manure, and groundwater over 4 years. The dairy operation associated with the study field had an approved DNMP. Monitoring nitrogen dynamics over multiple years allowed variations in weather, manure application, and crop stage to be taken into account.

The primary technical objectives of the study were to:

- Closely track nitrogen dynamics and fate at a grass field overlying the SBA that receives routine manure applications:
  - Quantify the sources of nitrogen input and output to the study field on an annual basis.
  - Quantify changes in soil nitrate and groundwater quality conditions in the study field over time, with a particular emphasis during the period at the end of the growing season.
  - Compare soil and groundwater results to state guidelines and standards.
- Identify the key farm management and environmental factors that influence groundwater and soil nitrate conditions.
- Perform an annual nitrogen mass balance evaluation for the study field.
- Use the mass balance and soil nitrate sampling results to estimate the annual nitrogen residual mass available for leaching to groundwater.
- Evaluate the correlation of the nitrogen residual estimates to the groundwater results, and determine the reliability of these estimates for predicting groundwater quality responses to manure management practices.
- Evaluate if current guidelines for manure management and soil monitoring are adequately protective of groundwater.
- Evaluate the study findings in the context of the SBA as a whole.

Location Description

Setting

The study site is a 22-acre grass field located in northwestern Whatcom County, Washington about 3 miles north of the town of Lynden and 0.3 mile south of the Canada border (Figure 1). The site lies on the flat Lynden Terrace, a glacial outwash plain that slopes gradually southward to the Nooksack River. Bertrand Creek, a perennial tributary of the Nooksack River, lies about 200 feet west of the western boundary of the site. The site is drained by surface ditches and waterways. The site elevation is approximately 130 feet (NAVD88).

Dairy wastewater/nutrients (hereafter referred to as manure) are typically applied as fertilizer on grass and corn fields, which are in turn harvested for livestock feed. Approximately 11 to 14 million pounds of manure nitrogen were applied to fields in Whatcom County in 2010 (Prest, 2011). Larger amounts of manure were applied across the Sumas-Blaine Aquifer (SBA) over the past 40 years, when more dairy cows were present. However, it is not clear how the loading rate (lb/acre) over the aquifer 40 years ago compares with the current loading rate.

Berry growing is also widespread in Whatcom County, in particular raspberry production. Whatcom County produces 65% of the U.S. supply of red raspberries (Whatcom Farm Friends http://www.wcfarmfriends.com/go/doc/1579/181808/). Inorganic fertilizer, the main nitrogen source for berries, is easily leached if not taken up by the crop. Kutchta (2012) found that common irrigation practices for raspberries can result in a large portion of the inorganic fertilizer leaching below the root zone and into groundwater. Other crops grown in the area include blueberries, strawberries, seed potatoes, and nursery stock.

On the Canadian side of the aquifer, poultry production and berry crops are intensive agricultural activities. Poultry production includes land application of manure. Inorganic nitrogen is also applied to berries in British Columbia.

Manured dairy fields lie to the east and west of the study field. However, directly upgradient (north) of the field lies a residence on a 3.5-acre lot. Groundwater beneath another property just east of the upgradient residence may also flow towards the study site during portions of the year, depending on the groundwater flow direction. Both residences were constructed in the past 10 years and are served by on-site sewage systems.

Climate

The regional climate is humid maritime with mild temperatures and rather high precipitation due to proximity to the Pacific Ocean. The Cascade and Rocky Mountains east of the site protect the area from cold air that otherwise would blow down from Canada. The mountains cause moisture rising off the ocean to drop 32 inches/year of precipitation in the southwestern part of the SBA. Precipitation rates increase to 60 inches/year closer to the mountains in Abbotsford, British Columbia.

The closest official weather station is the Environment Canada station at the Abbotsford, B.C. Airport, 7 miles east and slightly north of the study site. Precipitation is known to increase from southwest to northeast across the region; therefore, we assumed that precipitation at Abbotsford would be higher than that at the study site. Annual precipitation for the 30-year period prior to the study, 1973 to 2003, at Abbotsford Airport was approximately 61 inches (http://climate.weather.gc.ca/prods_servs/cdn_climate_summary_e.html).

Roughly 70% of the annual precipitation in the area typically occurs from October through March, outside most of the typical growing season for crops (Kuipers et al., 2012; Cox and Kahle, 1999). Little rainfall occurs in the summer. Potential evaporation at the Abbotsford Airport is typically over twice the precipitation rate from June through August, indicating a seasonal deficit in the water balance (www.Farmwest.com). Where available, irrigation water is applied to crops in the summer.

Soils

Hale silt loam soil overlies the study site. Hale soils are part of the Lynden-Hale-Tromp grouping that overlies much of the SBA. The subsoil at the site (11-27 inches) is mottled, indicating periodic reducing conditions.

When not artificially drained, the rooting depth for crops in Hale soils is limited by a seasonal high water table of 1 to 2 feet (U.S. Soil Conservation Service, 1992). Other characteristics of Hale silt loam include (U.S. Soil Conservation Service, 1992):

- 5-foot depth.
- Moderate permeability in the top 16 inches (0.6 to 2.0 inches/hour) and very rapid below that (greater than 20 inches/hour).
- Clay content, 10-18%.
- Organic matter content, 3-9%.
- pH, 5.1 to 6.5.

Hydrogeology

Regional hydrogeology

The study site lies in the Fraser-Whatcom Lowlands, also referred to as the Lynden Terrace, a glacial outwash plain that slopes gently south toward the Nooksack River. Repeated glacial advances and retreats during Pleistocene times deposited 1,000-2,000 feet of sediments over the area (Figure 3). Outwash from the last glacial episode, the Sumas Stade of the Fraser Glaciation, left gravel and cobble deposits near the Canadian border. These deposits grade finer southward to sand and some clay layers in the Lynden area (Easterbrook, 1971).

During the past 10,000 years, the Nooksack and Sumas Rivers have eroded and reworked the glacial deposits, resulting in the current flat, terraced flood plain morphology. The river has redistributed both the glacial and alluvial material, leaving gravel deposits in upstream areas as well as sand and silt downstream.

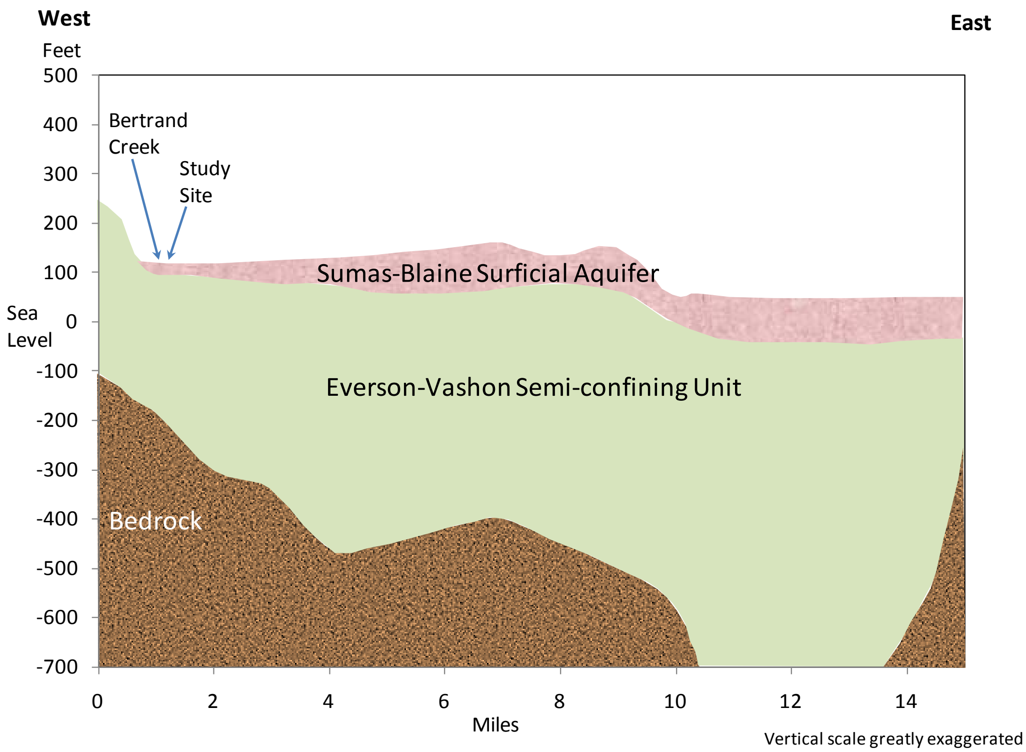

Figure 3. Generalized east-west cross-section near the study area.
Adapted from Cox and Kahle (1999) and Tooley and Erickson (1996).

The principal hydrogeologic units in the study area are shown in Figure 3 and include:

1. Sumas-Blaine Surficial Aquifer
2. Everson-Vashon Semiconfining Unit
3. Bedrock

Sumas-Blaine Aquifer (SBA)

The SBA is about 150 square miles in area and makes up the southern portion of the combined international aquifer system referred to as the Abbotsford-Sumas Aquifer (Figure 1). The SBA consists of stratified, unconsolidated sand and gravel outwash with minor clay lenses. The outwash grades from pebble-cobble alluvium just north of the Canada border in Abbotsford to sand with interbedded fine-grained lenses southwest of Lynden (Cox and Kahle, 1999).

The depth to water is less than 10 feet except for a small portion of the aquifer in the east, making it highly susceptible to surface contamination (Tooley and Erickson, 1996). A system of ditches and tile drains control high water table conditions and facilitate agricultural use in much of the area. Re-routing of a large portion of infiltrating water via tile drains prevents attenuation of leaching nitrate by denitrification and can quickly direct nitrate-rich leachate to surface water (Keller et al., 2008).

The regional groundwater flow direction is generally north to south in the northern part of the SBA (including the study site), toward the Nooksack River (Figure 1). However, local groundwater flow direction can vary (Tooley and Erickson, 1996; Graham, 2013).

The average saturated thickness of the SBA ranges from 25 feet near Blaine in the west to 75 feet near Sumas in the east, thinning at the margins of the alluvial plain (Figure 3). The study site is situated on SBA sediments at the northwestern margin of the plain (Figure 4, Plate 1).

Everson-Vashon semiconfining unit

The Everson-Vashon semiconfining unit is composed of glaciomarine drift consisting of unsorted pebbly clay and sandy silt (Cox and Kahle, 1999). This unit typically functions as a confining bed below the SBA but also includes local coarse-grained, water-bearing lenses as thick as 30 feet. The Everson-Vashon unit is typically 100 to 200 feet thick in the study area and 400 to 700 feet thick in the central axis of the aquifer (Figure 3). High groundwater ion concentrations and difficulty locating coarse-grained lenses preclude the Everson-Vashon unit from consideration as a reliable water supply. The confining layer also prevents significant transport of nitrogen to deeper zones.

Bedrock unit

The bedrock unit underlying the Everson-Vashon semiconfining unit consists of sandstone, mudstone, conglomerate, and coal of the Huntingdon and Chuckanut Formations (Figure 3) (Cox and Kahle, 1999; Creahan and Kelsey, 1988). This unit is not widely used for water supply due to depth and variable water-bearing properties. However, Cox and Kahle (1999) found records for 24 water wells that apparently connect with fractures where the unit is closer to the surface.

SBA properties

Hydraulic conductivity of the aquifer sediments varies widely over the SBA. Based on specific capacity estimates from driller's logs, Cox and Kahle (1999) reported horizontal hydraulic conductivity values in the SBA ranging between 7 and 7,800 feet/day, with a median value of 270 feet/day. Although hydraulic conductivity values varied dramatically over short distances, higher values tended to occur near the Canada border in the northeast part of the SBA. Lower values were measured in the western and southwestern parts of the aquifer. Site-specific measurements of hydraulic conductivity from study monitoring wells are discussed later in this report.

Cox and Kahle (1999) estimated horizontal groundwater velocity throughout the SBA at 0.2 to 29 feet/day based on specific capacity-derived hydraulic conductivity data for 218 wells. For most of the aquifer, they indicated that 2.5 feet/day is a reasonable estimate. Erickson (1991) estimated a groundwater velocity of 1-2 feet/day at a site 2 miles east of the study site using chloride as a tracer. Other velocity estimates for the SBA include 0.3 foot/day 1.8 miles southeast of the study site, based on short-term pumping test results at monitoring wells (Carey, 2002), and 25 feet/day in the coarser-grained Judson Lake area 7 miles east of the study site based on modeling results (Stasney, 2000).

SBA recharge

Recharge of water to the SBA is mainly from precipitation and occurs mostly from September through March, when precipitation exceeds evapotranspiration (Kuipers et al., 2012; Cox and Kahle, 1999).

Maps of recharge estimates by Cox and Kahle (1999) and Kohut (1987) show annual recharge estimates of 16 to 30 inches for most of the SBA, with increasing rates toward the north and east associated with higher precipitation. Recharge in the region is typically 60 to 80% of precipitation (Cox and Kahle, 1999; Malekani, 2012).

Large areas of the SBA are artificially drained to lower the water table below the root zone of crops, which prevents a portion of the infiltrating water from reaching the water table (Cox and Kahle, 1999). Drains typically operate during the winter and early spring, the time when most recharge occurs. The effect of the drain system on regional aquifer recharge rates has not been quantified. Tile drains are not present at the study site.

Study site hydrogeology

The study site is located on the western edge of the SBA, an area dominated by finer-grained material at the surface compared to most of the aquifer. The depth to the bottom of the aquifer is 40 feet at the site, based on well borings from the site (Appendix B, Well AKG726).

Figure 5 (Plate 1) shows the well locations used to develop hydrogeologic cross-sections for the site vicinity (Figure 6, Plate 1). Water level measurements shown on the cross-sections are from domestic and monitoring wells measured during different years; therefore, the water table position is an approximation. Where possible, low water table measurements for the fall were illustrated.

Surface water and groundwater from the site generally flow toward the Nooksack River, 5.5 miles south of the site. Also, localized seasonal reversal of the direction of the horizontal hydraulic gradient in the near vicinity of Bertrand Creek probably occurs, but was not observed in site monitoring wells.

Dairy field management

Regional practices and guidelines

Over the past 20 years, the method of applying manure to crop fields has changed at many farms from mainly large capacity “big gun” spraying and spreading with tanker trucks to methods that apply the manure closer to the soil surface for rapid infiltration or inject into the soil. These newer methods result in reduced loss of ammonia to volatilization and reduced odor, but potentially greater loss of nitrate to groundwater.

Dairies in western Washington typically begin applying manure to forage crops in the spring when weather and soil conditions are conducive to machinery traffic, crop uptake of nutrients is more active, and the risks of surface and groundwater contamination from bacteria and nitrate are reduced.

Non-application periods for grass fields in non-flood areas of Whatcom County are typically November 1 through February 15 or during periods when the T-Sum value is less than 200 (T-Sum 200).[6] In areas with potential flooding, the non-application period begins October 15 or 30 days prior to the typical flood season (November 15 for the Nooksack River).

The above ordinance also states that, "Should favorable climatic conditions exist, application may begin earlier in the spring than the dates established in this chapter, following approval from the Whatcom Conservation District Board based on T-Sum 200 or best available science. Soil conditions must also be considered when deciding when to apply nitrogen."

The Natural Resources Conservation Service (NRCS, 2005) provides guidance for nutrient management planners regarding conditions for winter manure application. The Whatcom Conservation District is testing a method for incorporating site-specific weather and field conditions into manure application timing and amounts (Application Risk Management) (Embertson, 2010).

Dairy producers may supplement manure applications with commercial inorganic nitrogen fertilizer. Irrigation water is applied on many fields during the dry summer months. Grass crops are typically harvested 4 to 5 times per year.

The treatment system used at the dairy (study site) did not include manure handling techniques to remove solids prior to lagoon storage, which influences the solids content of the manure.

Field management during the study

The 22-acre study field has received manure for over 20 years at about the same application rate as during the study (2004-2009), according to the dairy producer (~400 to 700 lb total N/acre/year). The site was planted in grass before the study and was tilled and re-seeded back to grass in April 2004 using conventional tillage practice. Conventional tillage practices include subsoiling, rototilling, plowing, disking, seedbed preparation, culti-mulching, and planting (VanWieringen and Harrison, 2009). The grass planted in 2004 consisted of 30 lbs Himark Fescue, 10 lbs Quartet Perennial Ryegrass, and 50 lbs Oats.

During the study, the dairy producer managed the field as before the study. The first liquid manure application for each year during the study occurred in February, March, or April, depending on weather and soil conditions. Manure was applied most often using an aerator (also referred to as subsurface deposition) with equipment from Aerway® Aerators & Parts (Figure 7). Tines were set 7.5 inches (19 cm) apart on a roller and allowed to drop 4 inches (10 cm) below the soil surface, creating intermittent slices 5 inches (12.5 cm) deep at the surface. Because tines

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[6] The T-Sum value is derived by summing the daily mean temperature (in degrees Celsius) starting January 1 of each year. (Ord. 98-074; Ord. 98-056--Whatcom County Code 16.28.030 www.codepublishing.com/wa/whatcomcounty/frameless/index.pl?path=../html/Whatco16/Whatco1628.html)

wear down over time and are not always fully inserted, slices may be less than 5 inches deep (Clark, 2013). Liquid manure was sprayed over the slices.

Figure 7. Sub-surface deposition of manure at the study site (top) and close-up diagram (bottom) from Aerway® Aerators & Parts.

Supplemental liquid manure from another dairy was applied by injection on 3 occasions in 2005 and 2008 (VanWieringen and Harrison, 2009). This method injects manure a few inches below the top of the soil.

Manure was typically applied 3 to 5 times per year following each grass cutting. The final manure application for any given study year occurred between the end of August and early October. In 2 of the 4 study years, the final manure application occurred after the last crop harvest (2005 and 2006).

Irrigation water from a nearby shallow well was applied at the study site using a hard-hose reel with a "big gun" sprinkler and pump each summer. The grass crop was harvested 4 to 5 times each year.

Study Design

Ecology's Environmental Assessment Program partnered with Washington State University's Livestock Nutrient Management Program (WSU) to accomplish the project objectives. We conducted the following tasks as part of a multi-media sampling program at the study site:

- Conducted a 4-½-year intensive multi-media monitoring program at a 22-acre manured dairy field.
- Analyzed the study field nitrogen mass balance each year and comparing the estimated nitrogen residual to shallow, underlying groundwater nitrate concentrations.
- Evaluated the effect of various environmental and management factors on the nitrogen mass balance and groundwater nitrate concentrations.

Our monitoring program focused on the following components of the nitrogen cycle to evaluate the balance of nitrogen at the study field system (crop, soil, and shallow groundwater) as shown in Figure 8:

- Inputs
  - Manure and inorganic fertilizer (mass of nitrogen applied to the field)
  - Irrigation water (volume and nitrogen concentration added to the field)
- Outputs
  - Grass harvested (mass of nitrogen removed from the field)
- Residual
  - Soil (fall nitrate concentration and estimated mass in the top 1-foot of soil)
  - Groundwater near the top of the water table (nitrogen concentration)

Nitrogen outputs due to volatilization were included in plant-available nitrogen estimates (Sullivan, 2008) or estimated using literature values. Denitrification in the soil was assumed to be negligible during the growing season (Sullivan, 2008).

To support the evaluation of nitrogen transport at the study site, additional field work was conducted to characterize the hydrogeology and soil characteristics of the study site. This included:

- Measuring static water levels in monitoring wells.
- Conducting tests of the hydraulic characteristics of the unconfined aquifer underlying the study field.
- Conducting grain size analysis of site deposits.
- Measuring chloride in groundwater to use as a conservative (non-reactive) tracer.
- Measuring other water quality constituents in groundwater that contribute to understanding nitrate levels.

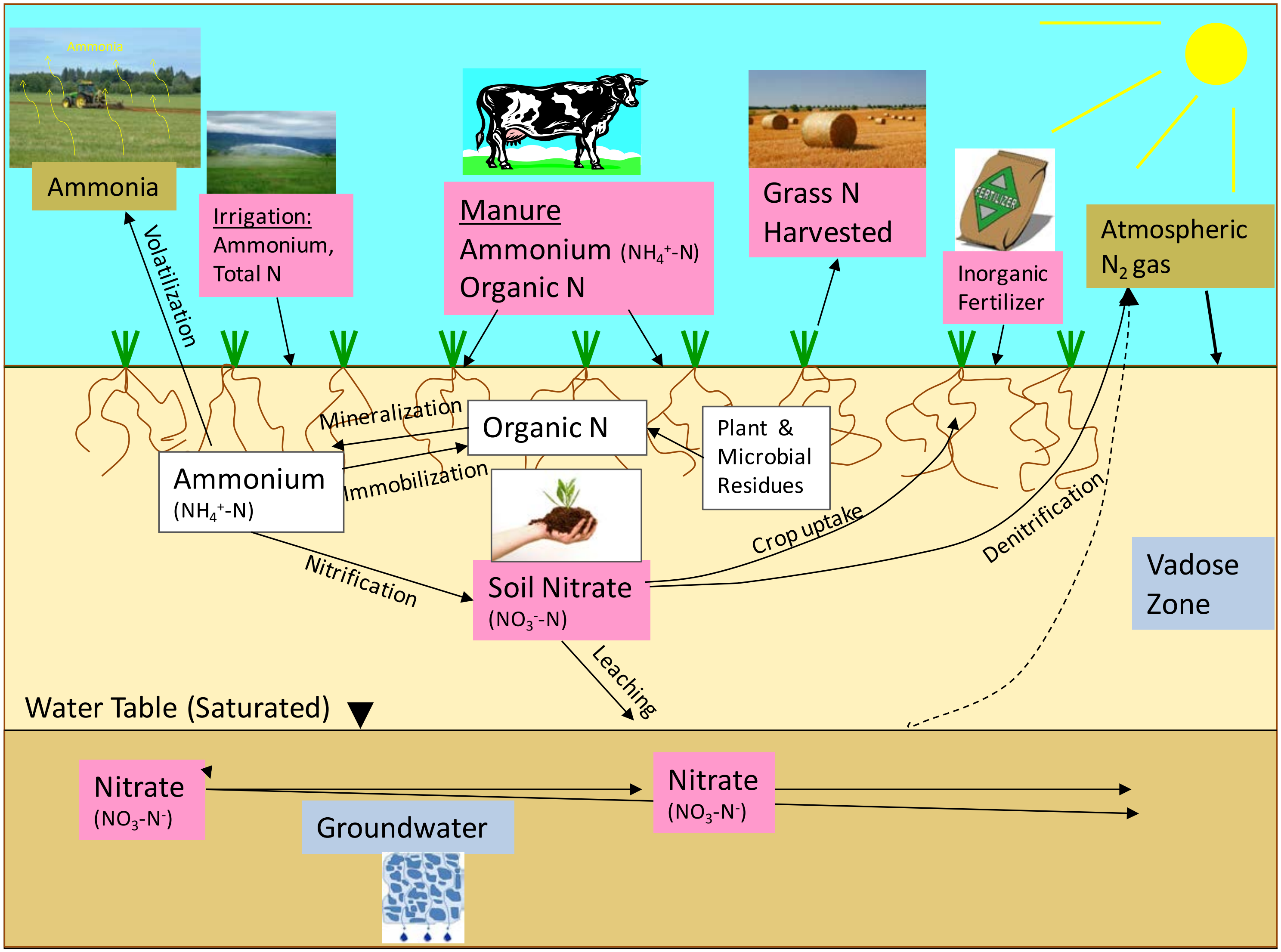

Figure 8. Major nitrogen compartments in the study field.

Media in pink boxes were monitored. Items in brown boxes were estimated.

Further details of the study design are described below and in Carey (2004). Figure 9 shows the analytes sampled for each media. Table 1 lists the analytes sampled and the frequency of sampling.

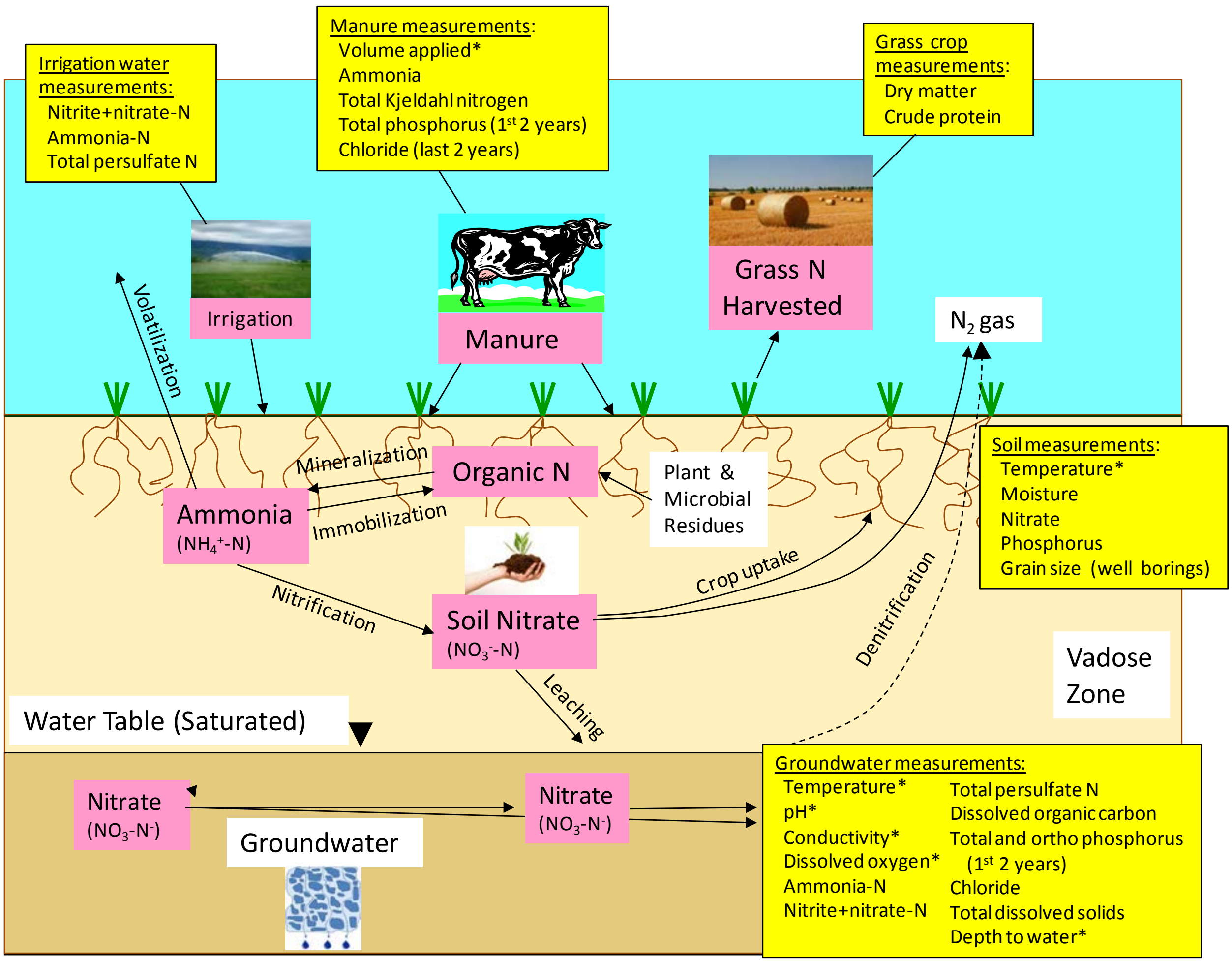

Figure 9. Analytes measured in each media are shown in boxes.

*Analytes were measured in the laboratory, except for those analytes with *s, which were measured in the field.*

Table 1. Analytes measured in groundwater, soil, manure, grass, and irrigation water. *Groundwater samples for laboratory analysis were filtered in the field except for groundwater from the deep well, AKG726, which were not filtered. Analytical methods are listed in Carey (2004).*

Analyte	Matrix[1]	Frequency
Field		
Groundwater temperature	G, S	Monthly (summer every 6 weeks)
pH	G	Monthly (summer every 6 weeks)
Specific conductivity	G	Monthly (summer every 6 weeks)
Dissolved oxygen	G	Monthly (summer every 6 weeks)
Soil temperature	S	Monthly (weekly August-November)
Soil moisture	S	Monthly (weekly August-November)
Laboratory		
Ammonium-N	G, M, I	G—Monthly (summer every 6 weeks) M—Each time manure applied I—Each time irrigation water applied
Nitrite+nitrate-N (Nitrate-N)	G, I	G—Monthly (summer every 6 weeks) I—Each time irrigation water applied
Nitrate-N	S	Weekly August-November, otherwise monthly
Total persulfate N	G, I	G—Monthly (summer every 6 weeks) I—Each time irrigation water applied
Total Kjeldahl N	M	Each time manure applied
Ortho phosphate	G	Monthly (summer every 6 weeks)—(2004-2006)
Total dissolved phosphorus	G, S, M	G—Monthly (summer every 6 weeks) S—Annually M—Each time manure applied
Chloride	G, M	G—Monthly (summer every 6 weeks) M—Each time manure applied
Total dissolved solids	G	Monthly (summer every 6 weeks)
Total organic carbon	G	G—Monthly (summer every 6 weeks)
Grain size	S	One time for drilling samples
Dry matter	Gs	Each time grass crop harvested
Crude protein (N)	Gs	Each time grass crop harvested
Organic matter	S	Annually
Soil chemistry[2]	S	Annually

[1] Matrix codes: G: Groundwater; S: Soil; M: Manure; Gs: Grass; I: Irrigation water.

[2] Soil chemistry: Phosphorus, potassium, boron, zinc, manganese, copper, iron, calcium, magnesium, sodium, sulfur, buffer pH, cation exchange capacity, total bases, base saturation, pH, electrical conductivity, ammonium N.

Methods

Weather conditions

A battery-powered Onset weather station was installed in the field during the study (Ecology weather station). Precipitation and air temperature measurements were recorded every 15 minutes from September 22, 2004 through March 18, 2009.

Nitrogen inputs and related constituents

Manure

Samples of liquid manure applied to the field were collected from the applicator when manure was being applied on 17 out of 18 times that it was applied. Manure was applied 4 to 5 times per year. The standard operating procedure (SOP) for manure sampling is described in Appendix C. The dairy producer reported the amount and timing of inorganic fertilizer application on 2 occasions, one time each in 2006 and 2007.

Irrigation water

Water for irrigation was applied each summer, with a total of 9 events during the study (Appendix D). Samples for volume and water quality were collected each time from the irrigation water applicator while the field was being irrigated, with the exception of 2 events in 2005. The amount of water applied during these 2 irrigation events was estimated by the producer.

Irrigation water samples were collected into 3 acid-washed buckets twice at different times throughout the irrigation event. The volume of water collected in each bucket was measured, and the rate of application was estimated using Equation 1:

$$I = \frac{\left[\frac{V}{T}\right]}{A} \qquad (Eq.\ 1)$$

where:
I = Irrigation rate (inches/day)
V = Volume of water in buckets (cubic inches)
T = Time (day)
A = Area of the buckets (square inches)

Contents of the 3 buckets were then composited and mixed in an acid-washed container. The sample was poured into 2 bottles with preservative and placed in a cooler with ice for shipping via FedEx to Ecology's Manchester Environmental Laboratory.

In addition to nitrite+nitrate -N, which was analyzed throughout the study, irrigation water samples collected on September 12, 2007 and for both applications in 2008 were also analyzed for ammonia-N and total persulfate N. “Nitrate+nitrite-N” is referred to as “nitrate-N” in this report, because nitrite-N is typically negligible in surface water and groundwater (Sawyer and McCarty, 1978).

Nitrogen outputs – grass crop

Grass samples were collected from 2-ft by 2-ft squares for yield estimate and quality analysis 1 to 2 days before each crop harvest, which occurred 4 to 5 times each year. Five subsamples were composited to form one sample on each date. The process was repeated for a duplicate sample.

The 10 general subsample locations (5 for the sample and 5 for the duplicate) were initially randomly selected and recorded using a Global Positioning System (GPS). The same 10 locations were then sampled each time thereafter. The SOP for grass sampling is described in Appendix E.

Soil conditions

Temperature

Soil temperature was also measured at a 6-inch depth during each soil sampling event. Soil temperature was measured by inserting 2 temperature probes 6 inches into the ground near the first soil coring location and averaging the 2 results.

Soil nitrate, moisture, organic matter, and other constituents

The frequency and timing of soil sampling events was designed to correspond with the likelihood of excess soil nitrate leaching to the water table. From December through July of each study year, when crops are growing most rapidly, precipitation rates are low, and reduced leaching of nitrate to groundwater is expected, we sampled soils on a monthly basis. From August through November, when the potential for leaching of residual nitrate from the soil column is of greatest concern (as precipitation begins to exceed evapotranspiration, and crop uptake rates declines), we sampled soils weekly.

The SOPs for soil sampling are summarized below and are based on methods described by Sullivan and Cogger (2003). The complete SOPs are described in Appendix F. Soil sampling locations are shown in Figure 10.

Each sample consisted of a composite of 15 soil core subsamples. A one-inch diameter hand-held coring device was used to collect each 1-foot deep soil core subsample at initially random locations around the field. The location of each core was verified using a GPS. The same locations were re-visited and sampled each subsequent event.

Loose crop or manure residue at the top of each core was discarded. The remaining soil from each of the 15 cores was placed in a 5-gallon bucket and mixed thoroughly by hand with a properly decontaminated trowel. The composite sample was then divided into 2 to 3 subsamples and placed in clean plastic bags, one for analysis at the contract laboratory, one for archival storage at WSU-Puyallup. One composite split sample was sent to a contract lab each year.

Figure 10. Soil sampling locations.

All sample sites were in the study field, although the aerial photo does not provide this level of accuracy.

Two sets of soil cores at 15 sites were used to obtain 2 samples of composited soil on each sampling date. A duplicate set of soil cores was collected each sampling day at a different set of 15 locations than were initially randomly selected. The duplicate sample was handled the same as the first sample. Subsequent soil cores were collected within a few feet of the original 30 duplicate locations.

Groundwater conditions

Monitoring well installations

Nitrate enters groundwater beneath the study field via 3 major pathways:

- Leaching and infiltration of nitrate from overlying soils through the vadose zone.
- Direct dissolution of nitrate in soil when the water table rises and saturates the lower portion of the soil column.
- Lateral groundwater transport of nitrate from upgradient of the study field.

Because of the high solubility of nitrate, infiltrating water can rapidly transport dissolved nitrate through the root zone and eventually to the water table. This process is of particular concern during periods of heavy precipitation (fall/winter/early spring), when the water table rises 5 to 9 feet in elevation, minimizing the transport distance to groundwater. This leaves little chance for nitrate to remain in the thin unsaturated soil layer (Kowalenko, 1989 and 1987; Zebarth et al., 1996).

Because the most recently recharged groundwater is closest to the top of the water table (Figure 11; Wassenaar et al., 2006), we completed the monitoring wells in a manner to intersect the water table and characterize recent recharge.

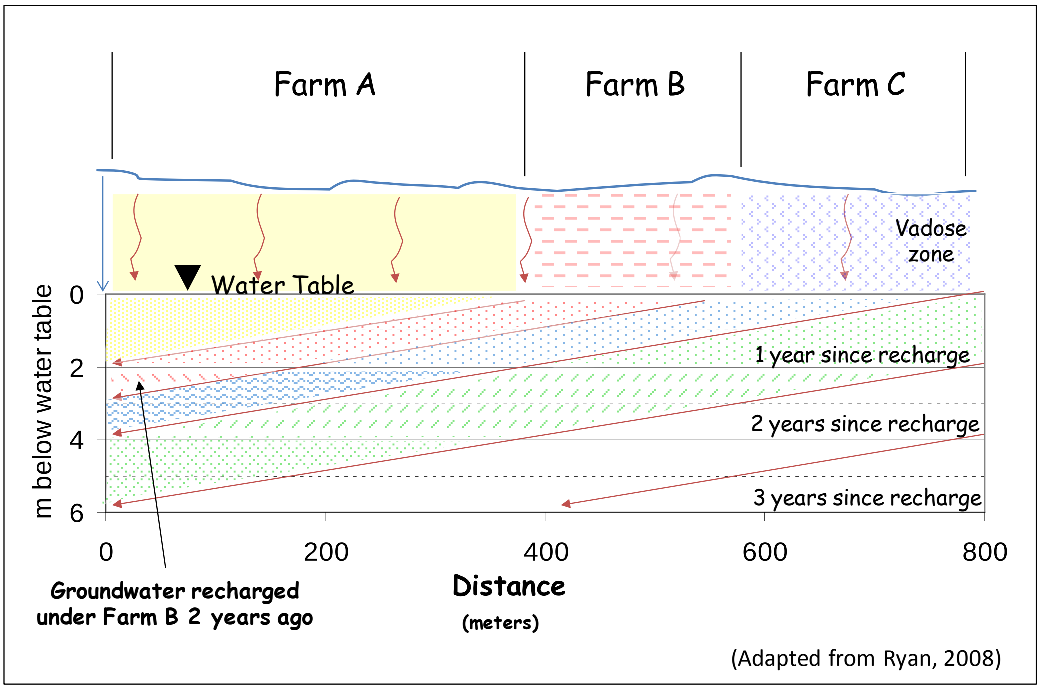

Figure 11. Generalized groundwater flow beneath fields in the Abbotsford area 7 miles northeast of the study site, showing that water near the top of the water table represents the most recent recharge from above.
Adapted from Ryan (2008).

The monitoring well network consisted of 6 shallow wells and 1 deep well in the study field. Two shallow wells were installed in 3 rows from upgradient to downgradient in the field (Figure 12). All monitoring wells were within the manured field (see Appendix B for drilling logs). The monitoring well locations and construction specifications were chosen to:

- Describe the subsurface hydrostratigraphy and hydraulic properties.
- Estimate the groundwater flow rate and direction.
- Obtain samples representative of the most recent groundwater entering the aquifer on the site (top of the water table).

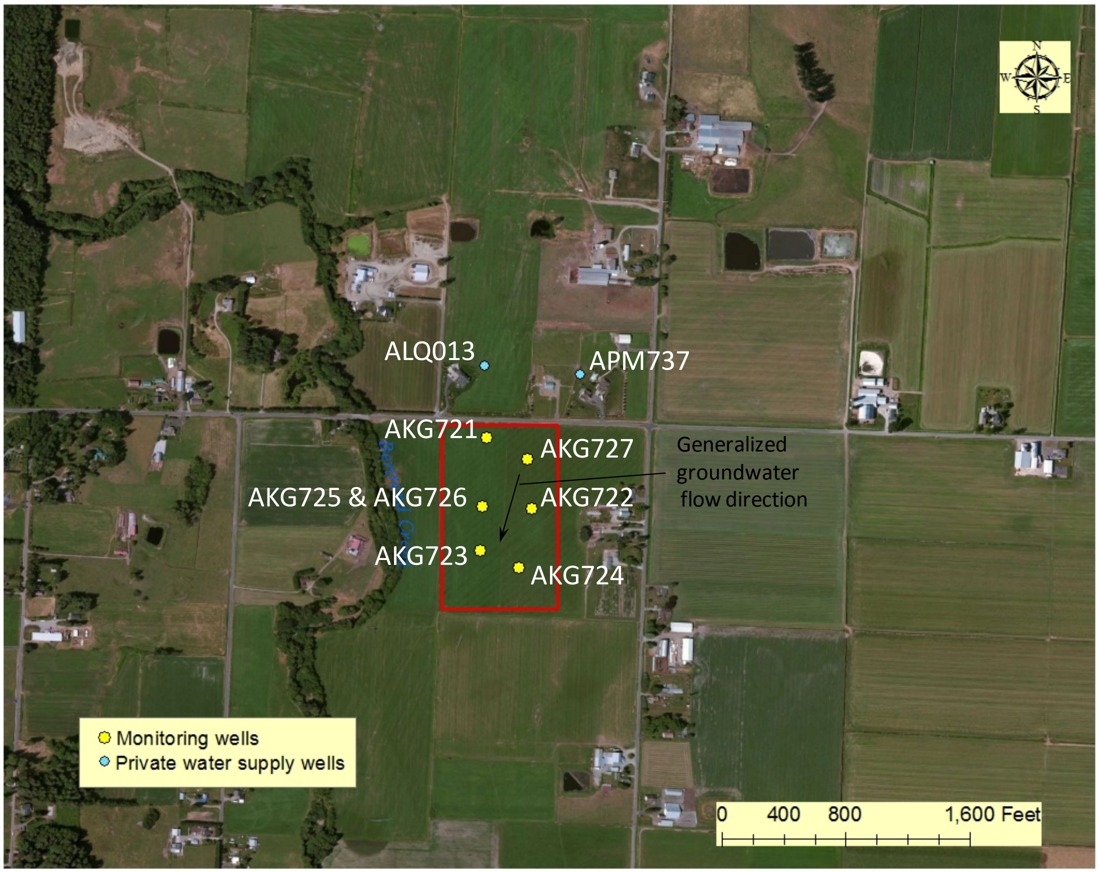

Figure 12. Location of wells sampled during the study.
All wells are in the study field.

Monitoring wells were installed by Holt Drilling, Inc., Puyallup, Washington, using a 4¼-inch inside-diameter hollow stem auger (8-inch outside-diameter). The wells were installed from August 25 to 26, 2004, about 4 weeks before groundwater samples were collected. Six wells were 12 to 13 feet deep, and one well drilled to the bottom of the unconfined aquifer was 38 feet deep. See Appendix G for a summary of well locations and construction information.

The monitoring wells were constructed according to the state standards for resource protection wells.[7] Figure 13 shows the standard construction plan for the shallow monitoring wells. The deep well was constructed similar to the Figure 13 plan except that the well screen was 10 feet long. The depth to the top of the screen was approximately 5 feet below ground surface in the shallow wells and 28 feet in the deep well.

Shallow wells were constructed with 2-inch inner diameter polyvinyl chloride (PVC), flush-threaded casing, and commercially fabricated 7-foot long screens (10-foot for the deep well) with a slot size of 20. We selected 7-foot long screens to provide as close to year-round access as possible to the top of the water table, which fluctuates roughly 7 feet over the year.

[7] Chapter 173-600 WAC

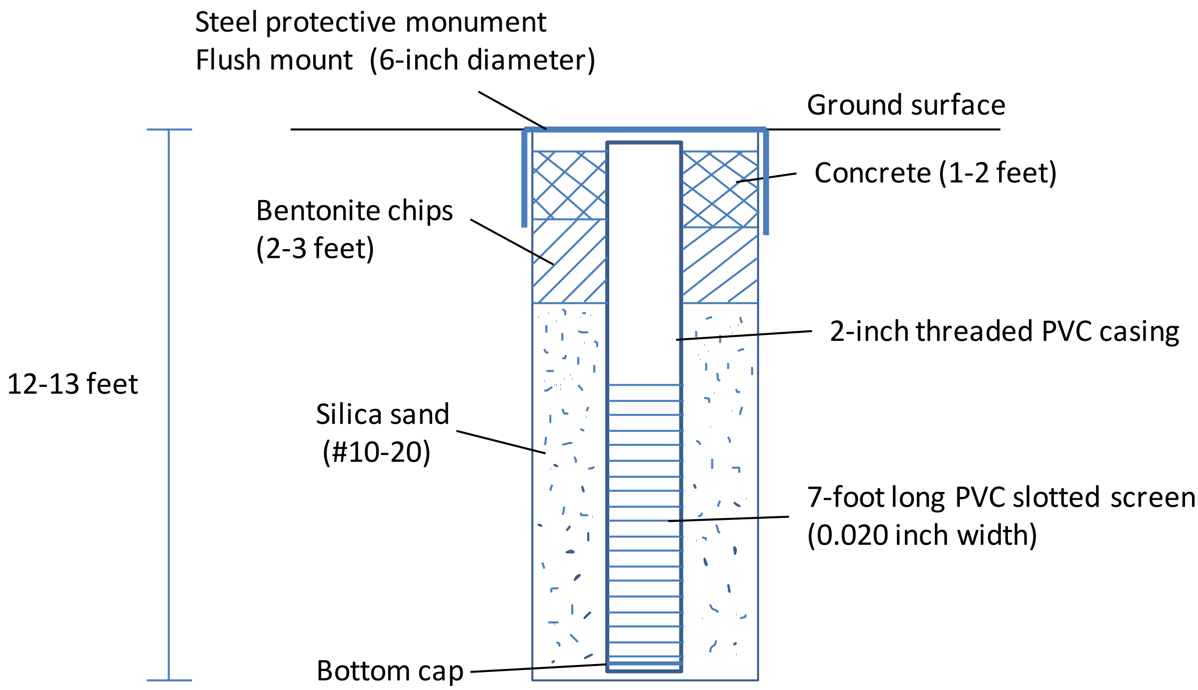

Figure 13. Schematic for shallow monitoring well construction.

The sand pack consisted of 10-20 silica sand installed continuously over the screened interval to 1 to 2 feet above the top of the screen. Bentonite pellets were placed within the annular space between the boring and the PVC casing from the top of the sand pack to within 1 to 2 feet of the surface. Concrete was installed around the top 1 to 2 feet of casing.

Split spoon core samples (18 inches long) were collected at 5-foot intervals during drilling. Core samples were placed in clean, labeled, plastic zip-lock bags. Fifteen split spoon samples were analyzed from the 7 wells for grain size according to ASTM Method D422 (ASTM, 2003). Sample intervals were selected to cover the range of material types encountered and the range of depths. Triplicate samples were analyzed for the deepest sample, 40 feet, in AKG726. The texture of the 40-foot sample changed dramatically from fine sand above 40 feet to clay and silt, indicating the base of the Sumas-Blaine Aquifer.

Each monitoring well was equipped with a water-tight cap and lock. A steel 6-inch diameter flush-mount outer protective casing was installed over the PVC well. The steel casing extended to a depth of 2 feet below ground.

After completion, the wells were developed by the driller using a jetting technique until the water removed from the borehole was free of sediment. A state well tag with a unique ID number was attached to each well.

Upgradient private wells

In addition to groundwater monitoring wells installed in the manured field, we sampled 2 upgradient private water-supply wells in 2008, one north of the study site and one northeast of the site (Figure 12). Both private wells are roughly 380 feet from the boundary of the study site.

Well ALQ013 was sampled 2 times (March 11 and April 2) and well APM737 one time on March 11 (see Appendix B for drilling logs). The wells are screened at 29 to 34 feet depth. Analytes included temperature, pH, specific conductivity, dissolved oxygen, ammonia-N, nitrate-N, total nitrogen, chloride, and total dissolved solids. Samples from these wells were not filtered. See Marti (2011a) for the SOP used to sample upgradient wells.

Hydraulic testing

We conducted aquifer hydraulic testing to determine if the subsurface hydraulic properties at the study site are similar to those reported for the Sumas-Blaine Aquifer as a whole. Hydraulic testing helps estimate sediment permeability and groundwater velocity, which affect how quickly nitrate and other dissolved constituents move once they reach the water table.

Short-term specific capacity tests were conducted on April 4, 2006 at 3 monitoring wells to provide an approximate estimate of the hydraulic properties of the aquifer materials. We tested the deep well (AKG726) and 2 shallow wells (AKG723 and AKG725) to characterize the shallow and deeper portions of the unconfined aquifer.

A specific capacity test consists of pumping a well at a known rate until the water level in the well equilibrates. The drawdown is recorded throughout the test period and is used with the well construction information to estimate the transmissivity and hydraulic conductivity of the aquifer in the vicinity of the well screen.

Specific capacity refers to the rate of well discharge divided by the drawdown in the well and is measured in gallons per minute per foot of drawdown. Bradbury and Rothschild (1985) developed a technique for estimating hydraulic conductivity using specific capacity based on the Theis (1963) graphical method. The Bradbury and Rothschild method uses a computerized iterative procedure to estimate transmissivity, which is then converted to hydraulic conductivity by integrating over the saturated thickness. The method uses the Cooper-Jacob approximation of the Theis equation with corrections for partial penetration and well loss (turbulent flow in the well during the test). See Appendix H for method details.

The assumptions of the Bradbury and Rothschild (1985) technique include:

- Confined, non-leaky, homogeneous, isotropic aquifer
- Storage coefficient is known
- Minimal well loss
- Penetration of the aquifer is known

Despite not meeting the assumptions of confined conditions, Bradbury and Rothschild (1985) had success using the computerized method in unconfined sand and gravel wells of the Central Sand Plain of Wisconsin. They found close agreement between results of full-scale pumping tests and specific capacity tests in individual wells. Sinclair (2002) likewise found close agreement between hydraulic conductivity results from large-scale aquifer tests in the Sequim-Dungeness area of Washington and results from specific capacity tests.

Groundwater sampling

The field and laboratory methods used for groundwater monitoring are described in Carey (2004). Standard protocols used in Ecology's Environmental Assessment Program were followed for measuring field parameters and collecting samples for laboratory analysis. Likewise, standard methods were used for sample handling, preservation, and storage (Marti, 2011b).

Groundwater samples were collected monthly during the fall and winter, and every 6 weeks in the spring and summer. Prior to sampling, water levels were measured at each well using a clean, calibrated electric probe per methods described by Marti (2009). Measurements were recorded to 0.01 foot and are considered accurate to 0.03 foot.

For well purge and sampling, we used a peristaltic pump with dedicated high density polyethylene tubing that remained inside the well between sampling events. A short section of silastic tubing at the pump head was used for all shallow wells and was replaced for each new sampling event. The pumping rate for the shallow wells was approximately 0.11 gallon/minute. The intake for the sample tubing was set at 1.5 feet below the top of the water table, or at the top of the screened interval when the water table was above the screened interval.

We purged and sampled the deep well (AKG726) using a submersible pump with dedicated polyethylene tubing. The pumping rate for purging and sampling at well AKG726 was approximately 1 gallon/minute.

We purged each shallow well for a minimum of 20 minutes and until field parameters (temperature, pH, conductivity, and dissolved oxygen) stabilized to within 10% for consecutive measurements spaced 5 minutes apart. The deep well was purged for 3 to 5 minutes, because the flow rate was high enough that field parameters stabilized quickly. Field parameters were measured inside an enclosed flow-through cell to minimize atmospheric bias effects (Figure 14).

Samples from the shallow wells were field-filtered using dedicated, in-line 0.45 um filters. After discarding the initial 50 milliliters of filtrate, samples were collected in clean bottles obtained from Ecology's Manchester Environmental Laboratory, as shown in Figure 15.

Samples collected from the deep well were not filtered. The higher discharge rate from the submersible pump used for the deep well made it more difficult to use field-filters. The discharge from the deep well was visually clear, and we assumed that the constituents of interest would not be in the particulate form.

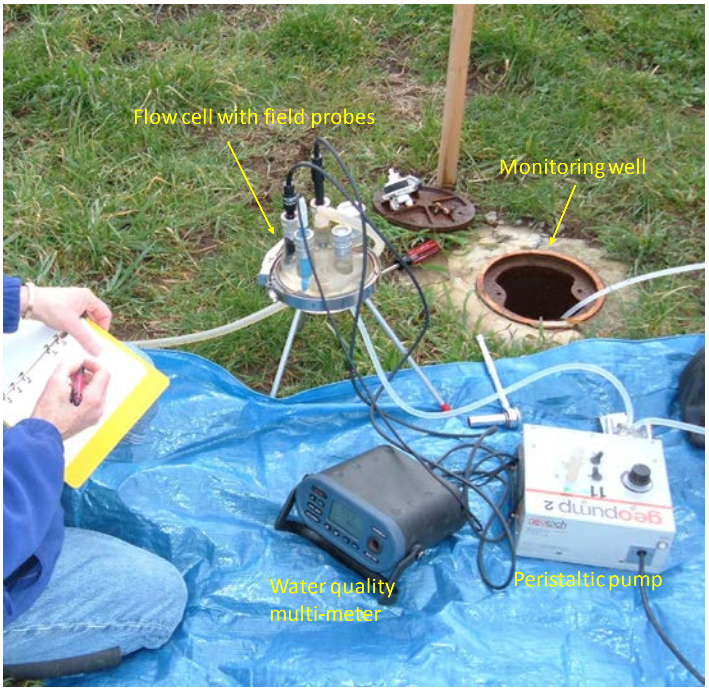

Figure 14. Groundwater sampling flow cell, peristaltic pump, and multi-meter for measuring field parameters: temperature, pH, conductivity, and dissolved oxygen.

Figure 15. Field-filtering a groundwater sample using a disposable, in-line filter that bypassed the flow cell.

Project Quality Assurance

Results of quality assurance testing for each media sampled are described in detail in Appendix I and summarized below. Overall, the results of the quality assurance testing indicated that the analytical data collected during the study are of good quality and can be used without qualification. In a few cases, qualifiers were added to a data result to identify values that may be outside of the project data quality objectives.

Manure

Duplicate manure samples were collected at least once each year and analyzed for percent solids (dry matter content), ammonium, and total nitrogen. See Appendix I, Table I.1, for results.

The range of relative standard deviations (RSD) for ammonium in duplicate manure samples was 0.08 to 30.4% with a mean of 8.0%. The project data quality objective RSD of 7% established in Carey (2004) was met on 4 out of 6 occasions.

The range of RSD for duplicates of manure total nitrogen samples was 2.57 to 18.9% with a mean of 8.0%. The target RSD of 7% was met on 3 out of 5 occasions. Manure results for dates that did not meet the target precision are qualified in the results (Appendix L).

Grass crop

Duplicate grass samples were collected each time the field was harvested from July 17, 2005 through October 21, 2008 (Appendix I, Table I.2). Individual samples were collected for the 3 harvests prior to July 17, 2005.

Eighty-three percent of wet-weight RSD values were within the 10% target (Appendix I, Table I.3). Eighty-nine percent of the dry-weight values and all of the crude protein values were within the 10% target range. Values outside the target range are qualified in Appendix I, Table I.2, and in the results (Appendix N). Based on our experience, the 10% target for RSD of grass parameters is particularly stringent.

Soil

Split soil samples were collected quarterly except in 2007, when split samples were not collected. A split sample consisted of a portion of 1 of the 2 duplicate samples for a given date. The split samples were analyzed by Soiltest Farm Consultants in Moses Lake, Washington.

The RSDs for 14 split soil nitrate samples are shown in Table I.4. The mean RSD was 11%, above the target of 7% (Carey, 2004).

Results for duplicate soil nitrate samples are shown in Table I.5. The mean RSD for 107 duplicates was 13.1%. Thirty-nine percent of the RSDs for duplicate soil nitrate samples met the 7% target precision.

The target precision for soil nitrate set prior to the start of the study may have been unreasonably low. The average RSD for 13 soil nitrate studies at dairy farms conducted by Washington State University was 16% (Bary, 2010). The range in soil nitrate concentration values for most of these studies was less than that in the current study.

Because the target precision for soil nitrate was so much lower than the average RSD in similar studies, and the range of soil nitrate values in the our study was so wide, a more realistic threshold for acceptable precision is 20%.

Soil nitrate duplicate samples with RSDs less than 20% are considered acceptable for use without qualification (Bary, 2012). Twenty-four of 107 soil nitrate duplicate samples exceeded the 20% threshold for RSD and are qualified in Table I.5 and in the results (Appendix K).

Groundwater

Field quality assurance

All groundwater field meters were calibrated at the start of each day according to the manufacturer's instructions. Replicate field measurements were collected at one monitoring well for each sampling event (replicating a different well each round) to assess overall precision of field and laboratory results (including the environmental variability over a few minutes between samples). Replicate samples were collected by immediately repeating the normal sampling process at the chosen well. Replicate samples were submitted blind to the laboratory.

The relative percent difference (RPD) for field parameters excluding dissolved oxygen was 0.6-1.4% (Appendix I, Table I.6.). The RSD for dissolved oxygen, which was often in the 0-3 mg/L range, was 8.7% (Table I.7).

The RPD values for water quality parameters represent combined field and laboratory precision. The target precision for nutrients, 7% RSD, and for chloride and dissolved organic carbon, 10% RSD, were met in most cases. Values that did not meet the targets are qualified in Appendix I., Table I.6.

Mean RPDs for laboratory analytes based on field replicates ranged between 2.2 and 5.3%, excluding total dissolved phosphorus (including nitrate-N, total persulfate nitrogen, ortho-phosphorus, chloride, dissolved organic carbon, and total dissolved solids). See Table I.7. The mean RPD for total phosphorus was 17%. Total dissolved phosphorus is the only analyte that did not meet the target precision.

On 6 occasions in 2008, a blank sample of de-ionized water from Manchester Laboratory was collected using the same silastic tubing for the peristaltic pump that had been used for sampling the monitoring wells. Results of blank samples were used to evaluate potential cross-contamination between sample locations from the silastic tubing. Most of the blank results for the nitrogen series were below detection (Table I.8). On May 6 and June 19, 2008, both nitrate-N and total persulfate nitrogen (TPN) were detected at concentrations roughly 1% of sample values. These results indicate that using the same silastic tubing when purging and sampling each well (one new piece of silastic tubing for each sampling event) was not a significant cross-contamination source.

Laboratory quality assurance

Laboratory quality assurance consisted of duplicate blanks, duplicate samples, spiked samples, and check (control) standards. Manchester Laboratory conducted internal quality assurance reviews. Most of the results are considered acceptable for use without qualification. Some data were qualified as described in Tables I.6 through I.8.

Impact of manure leakage on groundwater quality results

On February 7, 2005, the sealing well cap and flush-mount monument cover for monitoring well AKG727 were inadvertently left off the well. The field received one manure application before the error was discovered on March 3, 2005. The well was purged, resealed and remained closed for the rest of the study except when sampling. A 2-dimensional analytical model was used to estimate the potential impacts manure entry into the well had on the groundwater quality during the study (Appendix I.9).

The model assumed that manure was constantly injected into well AKG727 for 2 days. The resulting groundwater concentrations in well AKG727 and all downgradient wells were estimated using the model. Any estimated groundwater nitrate-N concentrations affected by more than 0.25 mg/L-N were disqualified. Only data from AKG727 were affected by 0.25 mg/L-N and greater. Potentially affected water quality data collected from February 7 through July 14, 2005 at AKG727 were not used in any data analyses in this report.

Results

Precipitation

The on-site weather station recorded precipitation and air temperature measurements on 75% of the days from September 22, 2004 through March 18, 2009 (1,262 days). Battery problems prevented data collection on 25% of days (418 days). Daily precipitation data at the study site for 1,072 days correlated with daily precipitation data from the Abbotsford, British Columbia Airport ($r^2 = 0.764$)[8]. The relationship between precipitation at the 2 sites was estimated using Equation 2:

$$y = 0.707\,x + 0.003 \qquad (Eq.\ 2)$$

where:
y = Precipitation at the study site (inches)
x = Precipitation at Abbotsford, B.C. Airport (inches)

Therefore, on dates when the weather station at the study site failed, daily precipitation was estimated as 70.7% of that at the Abbotsford Airport (Figure 16; monthly data are compiled in Appendix J, Table J.1, daily data in Table J.2). The annual precipitation at the study site using this method ranged from 40.0 inches in 2005 to 46.2 inches in 2007. The annual precipitation estimates are within 3 inches of the 30-year annual average of 43 inches (70.7% of the Abbotsford Airport 30-year average).

[8] January-April 2006 data were not included due to anomalies in Abbotsford Airport data.

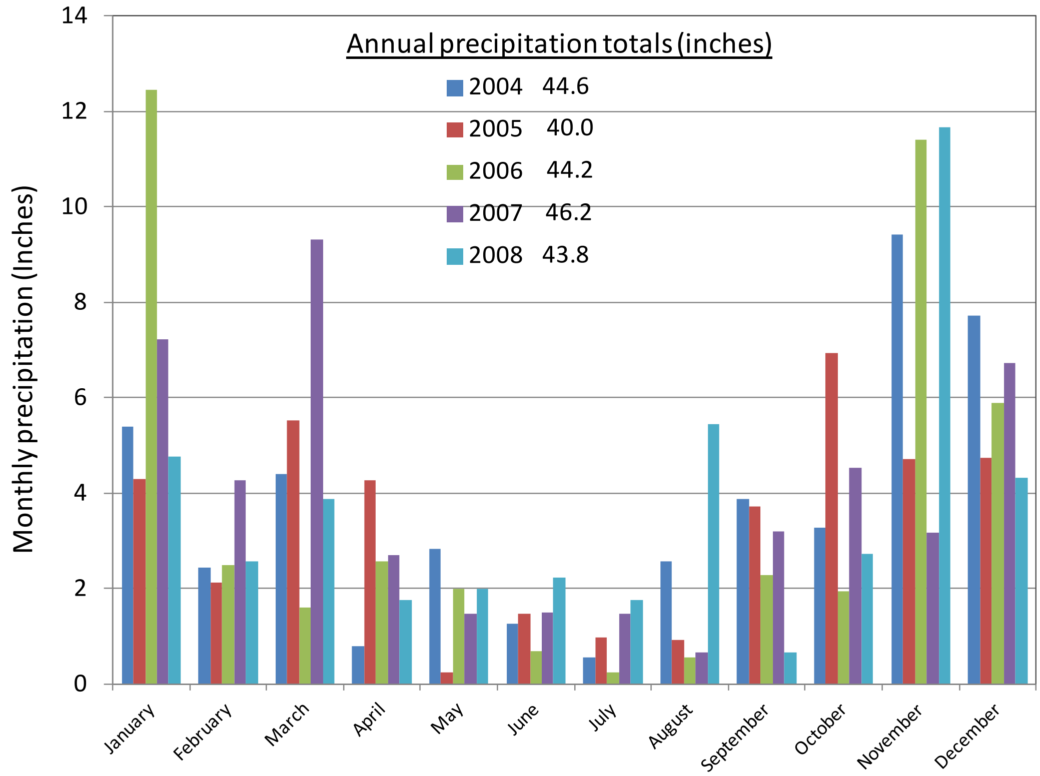

Figure 16. Monthly precipitation at the study site.

Data are from the on-site weather station for 75% of dates. Data for August and September 2004 (before the weather station was installed) and dates when the study site weather station was not operating were estimated as 70.7% of the Abbotsford, B.C. Airport values based on the data regression between the 2 sites.

Air temperature

The average annual daily temperature at the study site was 8.9° to 10.8° C (Table 2). Data from 2 nearby weather stations are included to fill data gaps when the on-site system failed: www.wunderground.com site KWALYNDE1 located between N. Pine Ct. and 19^{th} St. close to the intersection of Guide Meridian and Main St. and the WSU weather station in Lynden (Lat/Long: 49.00176/122.484523).

Table 2. Annual average, minimum, and maximum daily air temperature (ºC) at the study site.
Includes data from KWALYNDE1 and Washington State Univ.-Lynden weather stations when the on-site weather station was not operating. See Appendix J, Table J.3, for daily data.

Year	Average	Minimum	Maximum
2004	10.8	-8.9	25.0
2005	9.6	-6.0	20.3
2006	9.9	-10.2	23.9
2007	9.4	-7.3	25.7
2008	8.9	-9.8	23.0

Soil temperature and soil moisture

Soil temperatures ranged from -5.0° to 29.9 °C (Figure 17, Plate 2). (See Appendix K, Table K.1, for tabular data.) The 5 highest soil temperature measurements occurred in 2006, mostly in the late summer and fall.

Soil moisture measurements ranged from 12.7 to 54.7% (Figure 17, Plate 2). (See Table K.1 for tabular data.) The lowest soil moisture values occurred in the summers of 2005 and 2006. Below a soil moisture level of 20% of dry weight, grass crops commonly go dormant, resulting in little nitrogen uptake (VanWieringen and Harrison, 2009).

Soil temperature and soil moisture tended to be inversely related. When soil temperature was high in the summer, soil moisture tended to be low due to evapotranspiration. Heavy precipitation and low evapotranspiration in the winter kept the soil moist and cool.

On-site soil moisture results early in the study indicated that irrigation applied before the roots became dry could prevent the grass from going dormant in late summer. Therefore, the first application of water occurred earlier in the season each consecutive year in order to maintain grass growth during the dry late summer.

Soil organic matter and soil chemistry

Results for annual soil organic matter and soil chemistry sampling are shown in Appendix K, Table K.2. Soil organic matter ranged from 7.0 to 8.4%. The amount of organic nitrogen available for crops is sometimes calculated at 20 lbs/acre/year for each 1% organic matter, up to a total of 120 lb/acre (NRCS, 2006). The cation exchange capacity of the soil ranged from 19 to 23 milliequivalents/100 g.

Nitrogen and chloride inputs

Nitrogen inputs – manure and inorganic fertilizer

Manure made up the bulk of the total nitrogen mass applied to the field during the study (Figure 18). The timing and quantity of total nitrogen applied as manure (both organic nitrogen and ammonium-nitrogen) and as inorganic fertilizer are shown, by event, on Figure 19 (Plate 2). Inorganic fertilizer was applied only 2 times, once in 2006 and once in 2007 (31 and 48 lb/acre respectively). Atmospheric input was assumed to be 8 lb/acre (Kuipers et al., 2012). The nitrogen contribution from soil organic matter was not measured during the study.

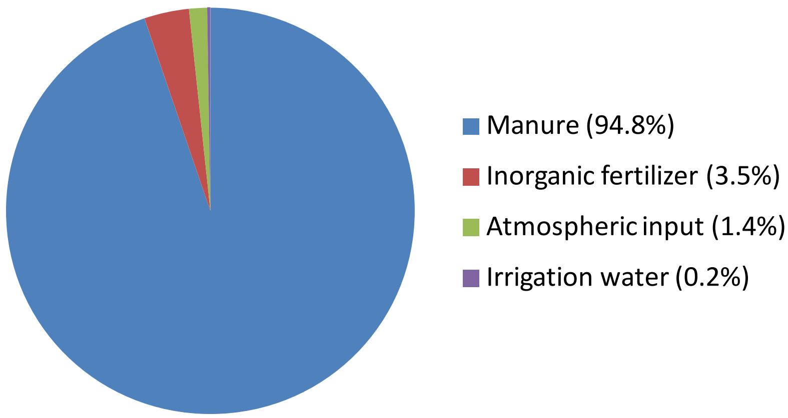

Figure 18. Nitrogen mass inputs to the ground surface by source for 2005 through 2008.
Atmospheric input is from Kuipers et al. (2012).

Manure was first applied each year in the spring. The earliest initial application was on February 18, 2005; the latest on April 27, 2006. The earliest final application for the year occurred on August 31, 2005 and the latest on October 5, 2006. Manure monitoring results are shown in tabular form in Appendix L, Table L.1.

The annual amount of total nitrogen applied by the producer to the field ranged from 394 to 715 lb/acre (Figure 20, Plate 2) with a mean of 548 lb/acre. Between 2005 and 2008, the average nitrogen composition of the applied manure was 47% ammonium-nitrogen and 53% organic nitrogen. The percentage of ammonium was highest in 2005 (67%) and lowest in 2007 (36%) (Figure 19, Plate 2). Nitrate was not measured in manure, because it is typically not a significant component (Beegle et al., 2008).

Most of the nitrogen was applied during the growing season, when there was little if any recharge and uptake by the crop was high. However, depending on the year, between 14 to 25% of nitrogen was applied between October and March (90 to 170 lb/acre/year) when groundwater recharge increases and crop uptake is low.

The average nitrogen content in manure, 15 lb/1,000 gallons, was in the high range compared with manure from 25 Whatcom County dairies with similar solids content (3.5%) reported by Sullivan et al. (1994) (Appendix L, Table L.2). The regression in Sullivan et al. (1994) indicated a total nitrogen content of 10.1 to 12.3 lb/1,000 gallons for manure with 3 to 4% solids. Solids and nitrogen content in manure vary depending on the type of treatment system used prior to field application. Dairies with flush systems, second stage lagoons, and mechanical solids separators typically have lower solids content. However, the dairy where the manure was produced in the current study did not have these systems.

Nitrogen inputs – irrigation water

Annual irrigation water volume totals ranged from 2.5 to 5.7 inches of water (66,000 to 155,000 gallons/acre) as shown in Table 3.

Table 3. Schedule of irrigation water applied.

Year	Number of irrigation events	1st application date	1st application amount (inches/acre)	2nd application date	2nd application amount (inches/acre)	3rd application date	3rd application amount (inches/acre)	Total (inches/ acre)
2005[1]	2	9/15/2005	1.25	10/15/2005	1.25			2.50
2006	2	7/22/2006	1.75	8/22/2006	3.94			5.69
2007	3	7/17/2007	1.96	8/23/2007	1.95	9/12/2007	1.52	5.43
2008	2	7/8/2008	1.95	8/16/2008	2.46			4.41

[1] Dates and amounts are estimates.

Nitrate-N concentrations were measured each year in irrigation water (Figure 21, Plate 2). Ammonium-N and total persulfate nitrogen (TPN) concentrations were analyzed in 2007 and 2008 but not in 2005 and 2006. Because the same source of water was used in 2005 and 2006, and nitrate concentrations were similar in all years, we assumed that the average total nitrogen concentration for 2007 and 2008, 1.3 mg/L-N, was also representative of 2005 and 2006 (Appendix M, Table M.1). The total annual nitrogen mass input from irrigation, 0.8 to 1.8 lb/acre/year, was roughly 0.2% of the total nitrogen applied (Figure 18; Table M.2).

Chloride inputs

The mass of chloride applied during each manure application is shown in Figure 22 (Plate 2) (Appendix L, Table L.2, for data). Chloride was not measured in irrigation water and inorganic fertilizer and is assumed to be negligible in both. The average rate of chloride application to the field in manure was 40 lb/acre/year. The annual total mass of chloride applied was 111 to 205 lb/acre (Figure 23, Plate 2). The application rate for chloride for individual manure applications was correlated with the application rate for ammonium-nitrogen, the most available form of nitrogen in manure (r^2=0.60, n=17).

Nitrogen outputs

Grass crop

The total nitrogen harvested for each grass cutting event is shown in Figure 24 (Plate 2) and was calculated using Equation 3:

$$TN = \left[\frac{CP}{6.25}\right] x\ DM \qquad (Eq.\ 3)$$

where:
TN = Total nitrogen (lb/acre)
CP = Crude protein (%)
DM = Dry matter (lb/acre)
(See Appendix N for results of crude protein, dry matter and total nitrogen removed in the crop.)

Figure 25 (Plate 2) shows the annual totals of nitrogen harvested. The highest annual total nitrogen yield occurred in 2007 (457 lb/acre), when the total nitrogen applied was 434 lb/acre. The lowest annual nitrogen uptake, 393 lb/acre, occurred in 2008, the year with the highest amount of nitrogen applied (715 lb/acre).

The estimate for 2005 nitrogen harvested includes results for the last grass crop of the year (102 lb/acre), which was not actually removed from the field due to inclement weather. This unrealized harvest was included to represent the uptake of nitrogen for the year, even though it was not removed from the field. This biased the 2005 crop uptake estimate high, because some of the grass that was not removed decomposed and became available for leaching.

Nitrogen residual – soil nitrate

Soil nitrate results are shown in Figure 26 (Plate 2; data in Appendix K, Table K.1). The range for soil nitrate concentrations (August through November) was 5.5 to 60 mg/kg.

Equation 4 was used to convert mg/kg dry weight of soil nitrate to lb/acre. The bulk density value for the Hale silt loam soil (0- to 10-inch depth) at the site is 1.13, assuming the average organic matter content at the site = 7.5%. Therefore, the conversion factor from mg/kg nitrate dry weight to lb/acre ($\rho_b\ x\ 2.791$) is 3.1. This is somewhat lower than the suggested conversion for western Washington, 3.5, in Sullivan and Cogger (2003).

$$N = C_{Soil\ N}\ x\ \ \rho_b\ x\ 2.79 \qquad (Eq.\ 4)$$

where:
N = Soil nitrate concentration (lb/acre)
$C_{Soil\ N}$ = Soil nitrate concentration, dry weight (mg/kg DW)
ρ_b = Soil dry bulk density (g/cm^3)
2.791 = Unit conversion constant [to convert mg nitrate-N/kg (dry weight) to lbs nitrate /acre-ft]

The data results indicate:

- The range of soil nitrate concentrations observed during the September to October period, when post-harvest soil nitrate samples are usually collected (Plate 2, Figure 26, green boxes), was 11.5 to 43 mg/kg (36 to 133 lb/acre).
- The range of concentrations observed for the 10 samples that were collected on a date that met the guidance provided by Sullivan and Cogger (as soon as possible after the last manure application and before 5 inches of precipitation starting September 1; see Table 4) was 15.5 to 43 mg/kg (48 to 132 lbs/acre).
- If sample dates in November are included in the post-harvest period, the soil nitrate concentration observed ranged from 5.5 to 60 mg/kg (17 to 186 lbs/acre).
- The highest value observed, 60 mg/kg (186 lb/acre) on November 8, 2006, was outside of the recommended sampling period.
- Soil nitrate concentrations were highly variable. There was typically a 2-fold difference between the maximum and minimum weekly soil nitrate concentrations in the fall season each year (September 1 through October 31), or up to 24 mg/kg (74 lb/acre). Temperature, precipitation, spatial heterogeneity, timing of the last manure application, and other factors influence these changes.

Table 4. Post-harvest soil nitrate results that met the timing protocols recommended in Cogger and Sullivan (2003).

Date	Soil nitrate concentration (mg/kg)	Soil nitrate mass (lb/acre)
9/9/2004	43.0	133
9/17/2004	28.5	88
10/1/2004	19.0	59
10/4/2005	15.5[1]	48
10/11/2005	16.0[1]	50
10/9/2007	16.9	52
10/16/2007	18.8	58
10/17/2008	26.9	83
10/24/2008	30.8	95
10/31/2008	29.9	93

[1] Last manure application occurred after the last harvest.

Groundwater conditions

Hydrogeologic conditions

Aquifer properties

Grain size distribution

Split-spoon soil samples from monitoring well borings analyzed for grain size distribution were used to classify soil samples according to ASTM Method 2487-92 (ASTM, 1994). Table 5 lists the values for effective grain size (d_{10}), uniformity coefficient (C_u), and coefficient of curvature (C_c). These values were calculated using particle size distribution curves and were used to classify soils. Soil classification results are shown in cross-section in Figure 27 (Plate 1). See Appendix O for particle size distribution curves.

The effective grain size, d_{10}, is used here to qualitatively compare the potential rate of leaching and the potential for denitrification. The d_{10} represents the sieve diameter through which only the smallest 10% of the particles pass. The lower the d_{10} value, the larger the portion of fine-grained material in the sample. Slow percolation of liquid through fine-grained material allows for bacterial or chemical processes that use up oxygen and enhance denitrification potential if there is a sufficient electron source such as organic carbon.

The uppermost sediments varied among fine-grained classifications of clay or silt with sand and sand with silt and clay. At side-by-side borings, AKG725 and AKG726, samples from 7.5 to 25 feet below ground surface (BGS) contained little silt or clay and were categorized as well graded sand. Samples below 7.5 feet in the other wells had varying amounts of fine-grained material. But samples from all wells indicated more rapid movement of water vertically and horizontally below 7.5 feet than at shallower depths. The deepest sample was collected from the top of the confining layer that forms the base of the aquifer at 40 feet (AKG726).

Table 5. Particle size distributions for split spoon soil samples collected during installation of monitoring wells.

Well	Depth (feet BGS)	Soil class[1]	Description	d_{10} (mm)[2]	C_u[3]	C_c[4]	D_{60}	D_{10}	D_{30}
AKG-721	2.5	SM or SC	Sand with silt or clay	<0.075	166.7	4.8	0.500	0.003	0.085
AKG-721	5.0	ML with sand, or CL with sand	Silt or clay with sand	<0.001					
AKG-721	10.0	SW-SM or SW-SC	Well graded sand with silt or clay	0.078	3.2	1.3	0.270	0.085	0.170
AKG-722	5.0	SW-SM or SW-SC	Well graded sand with silt or clay	0.122	2.0	2.3	0.240	0.122	0.260
AKG-722	10.0	SW-SM or SW-SC	Well graded sand with silt or clay	0.115	2.4	0.9	0.280	0.115	0.170
AKG-723	2.5	CL or ML with sand, or CL-ML with sand	Clay or silt with sand or silty clay with sand	<0.001					
AKG-723	10.0	SP-SM or SP-SC	Poorly graded sand with silt or clay	0.087	4.4	2.2	0.380	0.087	0.270
AKG-724	7.5	SP	Poorly graded sand	0.169	4.4	1.4	0.750	0.169	0.420
AKG-725	2.5	CL or ML with sand, or CL-ML with sand	Clay or silt with sand or silty clay with sand	<0.075					
AKG-725	7.5	SW	Well graded sand	0.096	2.7	1.2	0.260	0.096	0.170
AKG-726	15.0	SW	Well graded sand	0.139	1.7	2.0	0.230	0.139	0.250
AKG-726	25.0	SW	Well graded sand	0.109	2.8	1.2	0.300	0.109	0.200
AKG-726	40.0	CL or ML with sand, or CL-ML with sand	Clay or silt with sand or silty clay with sand	<0.0013					
AKG-727	2.5	SM or SC	Sand with silt or clay	<0.0013	120.8	15.5	0.145	0.001	0.052
AKG-727	10.0	SW-SM or SW-SC	Well graded sand with silt or clay	0.087	3.2	1.3	0.260	0.082	0.165

[1] Plasticity index and liquid limit were not determined; therefore, silt and clay could not be distinguished. SW=Well-graded sand, SP=Poorly-graded sand, SM=Silty sand, SC=Clayey sand, CL=Clay, ML=Silt.
[2] Effective grain size: Particle size diameter through which 10% of sample particles pass on cumulative particle size distribution curve.
[3] C_u: D_{60}/D_{10} (Coefficient of Uniformity – if 1-3, then well graded; if greater than 3, then poorly graded).
[4] C_C: $(D_{30})^2/(D_{10} \times D_{60})$ (Coefficient of curvature measures the shape of the particle size curve indicating gradation).

Horizontal hydraulic conductivity

The Bradbury and Rothschild (1985) method was used to estimate horizontal hydraulic conductivity (K_H), a measure of the permeability of the aquifer sediments (see Appendix P for details). Hydraulic conductivity is used to estimate the velocity of groundwater flow.

Specific capacity results and estimated K_H values for 3 on-site monitoring wells are shown in Table 6. Two of the wells (AKG725 and AKG726) are only 3 feet apart but are screened at different depth intervals. Monitoring wells AKG725 and AKG723 are screened from 6 to 13 feet below ground surface (BGS); AKG726 from 25 to 35 feet BGS. The average K_H value at the site was approximately 53 feet/day.

Table 6. Horizontal hydraulic conductivity (K_H) estimates based on specific capacity.
Bradbury and Rothschild, 1985.

Well I.D.	Static water level (feet)	Pumping water level (feet)	Saturated screen length (feet)	Assumed storage coefficient	Aquifer thickness (feet)	K_H (feet/sec)	K_H (feet/day)
AKG726	8.77	8.98	10.0	0.20	35	7.95E-04	69
AKG725	8.72	8.78	4.3	0.20	35	6.07E-04	53
AKG723	7.98	8.06	4.7	0.20	35	4.17E-04	36

K_H: Horizontal hydraulic conductivity (rate of flow through a material over time at a unit gradient).

Groundwater occurrence and movement

Groundwater elevations

Hydrographs of water level elevations are shown in Figure 28 (Plate 3). The highest water levels occurred in the winter (December through March); the lowest in the fall (September through October). (See Appendix Q, Table Q.1, for data in tabular form.) On January 10 to 11, 2006, monitoring wells AKG722 and AKG727 were submerged and could not be monitored. Subsequent water quality and water level data did not indicate leakage from surface water to the well screen.

Depth to water

Depth to water from the top of the casing in the monitoring wells ranged from 0 to 11.4 feet (Figure 29, Plate 3). (See Table Q.2 for tabular data.) The shallowest water table values (0 to 5.2 feet BGS) occurred in winter months, coincident with the period of highest potential for nitrate leaching and the lowest potential for crop uptake of nutrients (December through March). The deepest annual water table depths, 10.4 to 11.4 feet, usually occurred in October.

The annual range of depth-to-water measurements in individual wells between highest and lowest depths was 4.5 to 10.1 feet/year. The mean annual difference between high and low water table depths was 7 feet.

Groundwater flow direction

Water level contours typical for fall low conditions (October 1, 2007), high winter conditions (December 28, 2004), and middle-range spring and summer conditions (March 2, 2005 and June 26, 2006) are shown in Figure 30 (Plate 3). The groundwater flow direction was consistently south-southwest toward the Nooksack River.

Hydraulic gradient

The horizontal hydraulic gradient ($i_{H;}$ dimensionless), or slope of the water table, was calculated using Equation 5:

$$i_H = \frac{dh}{dl} \qquad \textit{(Eq. 5)}$$

where:
dh = change in hydraulic head between 2 points (feet)
dl = lateral distance between 2 points (feet)

Hydraulic gradients for the contours in Figure 30 ranged from 0.0027 to 0.0051 with an average of 0.0037 (Table 7). Hydraulic gradients tended to be lowest in the late summer to fall season and highest during the winter.

Table 7. Horizontal hydraulic gradient estimates at the study site on 4 representative dates.
See Figure 30 (Plate 3).

Date	Horizontal hydraulic gradient (i_H)
12/28/2004	0.0051
3/2/2005	0.0037
6/26/2006	0.0033
10/1/2007	0.0027

Positive vertical hydraulic gradients were measured at the side-by-side shallow (13 feet deep) and deep (38 feet deep) wells, AKG725 and AKG726, indicating a downward hydraulic potential throughout the study period (Figure 31, Plate 3). An increasingly positive trend in the vertical hydraulic gradient value over the study period indicates that water and dissolved constituents moved downward more quickly over time. The mean vertical hydraulic gradient was 0.0047 (Standard deviation = 0.001, n=43).

Groundwater flow velocity

The average horizontal velocity of groundwater flow was estimated using a variation of Darcy's Law:

$$v = \frac{-K_H(\frac{dh}{dl})}{n_e} \qquad \textit{(Eq. 6)}$$

where:
v = Average linear groundwater velocity (feet/day)

K_H = Horizontal hydraulic conductivity (feet/day)
dh/dl = Horizontal hydraulic gradient (dimensionless)
n_e = Effective porosity (ratio of the volume of interconnected voids/volume of material that is capable of transmitting fluid)

We used the K_H value for well AKG725 (Table 8), because it is in the middle of the field and probably most representative of the shallow water table at the site. The annual range of horizontal hydraulic gradient was used for dh/dl.

The range of groundwater velocity estimates was 0.57 to 1.08 feet/day, or 208 to 394 feet/year, with a mean value of 0.78 foot/day, or 284 feet/year (Table 8).

Table 8. Estimates of groundwater velocity at the study site using the minimum, average, and maximum hydraulic gradients for dates shown in Table 7.
See Figure 30 (Plate 3).

Hydraulic gradient	Hydraulic conductivity (K_H)[1] (feet/day)	Hydraulic gradient (feet/feet)	Effective porosity[2]	Velocity (feet/day)
Minimum	53	0.0027	0.25	0.57
Average	53	0.0037	0.25	0.78
Maximum	53	0.0051	0.25	1.08

[1] From Table 6.
[2] Representative value for glacial outwash aquifers.

Recharge

Recharge was estimated using the water balance method shown in Equation 7 (Healy and Scanlon, 2010; Malekani, 2012):

$$R = P - PET \qquad (Eq.\ 7)$$

where:
R = Monthly recharge (inches)
P = Monthly precipitation (inches)
PET = Monthly potential evapotranspiration (inches)

Monthly potential evapotranspiration data were drawn from estimates developed by Environment Canada for the Abbotsford Airport climate station. These estimates were calculated using a modification of the Penman Monteith Equation for a grass crop (www.farmwest.com/node/930). During months when PET exceeded precipitation (generally May through August), recharge was assigned a value of 0 inches.

Annual recharge estimates ranged from 23.5 to 29.4 inches and are shown on Figure 31 (see Appendix J, Tables J.4 and J.5, for tabular data). Recharge comprised an average of 62% of annual precipitation, when annual precipitation was calculated from September 1 to August 31.

This calculation period is similar to the water year used in hydrologic budgeting (October 1 to September 30) but shifted back one month to fit climate patterns for the area and nitrate leaching potential. An average of 65% of the annual recharge occurred from October through March. This is similar to previous reports (Kuipers et al., 2012).

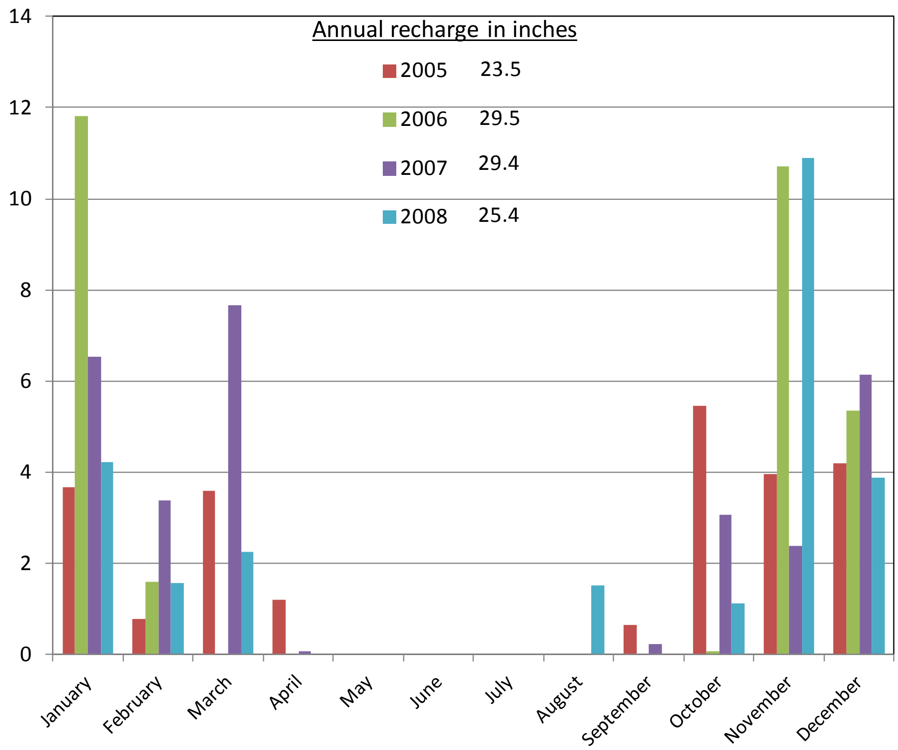

Figure 32. Estimated monthly recharge for the study site in inches.

Most of the focus on nitrate loss to groundwater is on the fall/early winter season; however, recharge and associated leaching continues to occur in late winter/early spring when precipitation exceeds evapotranspiration (Zebarth, 2013; Kuipers et al., 2012; Chesnaux and Allen, 2007).

Groundwater quality conditions

Time series results for groundwater quality are shown in Plates 4 and 5. Results are shown for pH, dissolved oxygen, conductivity, nitrate-N, chloride, organic carbon, total dissolved solids, and total phosphorus. Total nitrogen concentrations were similar to nitrate-N and are not shown graphically. Samples collected at well AKG727 between February 7 and July 14, 2005 were rejected as described in the *Quality Assurance* section and are not included in the results summary. See Appendix R, Table R.1, for monitoring well groundwater quality data in tabular form. Table R.2 summarizes results from upgradient private wells.

pH

pH affects the oxidation/reduction state of ammonia in groundwater. When pH is below 8, most of the ammonia is in the ammonium form (NH_4^+). This is the case in most of western Washington groundwater and surface water. pH also affects the rate of bacterial conversions of ammonia to nitrate (nitrification) and nitrate to nitrogen gas (denitrification) (Buss et al., 2004 and 2005; Coyne, 2008). (See *Nitrogen Cycle* section above.)

Results for groundwater pH shown in Figure 33 (Plate 4) were all below 8, indicating that ammonium was the predominant form of ammonia in groundwater at the study site. pH values ranged from 5.1 to 6.0 in all monitoring wells except AKG724 and AKG726 and did not vary significantly seasonally. The lowest pH values occurred in well AKG724, where values were consistently below 5.0. The highest pH occurred in the deep well, AKG726, with values typically around 6.5.

Dissolved oxygen (DO)

Results for DO are shown in Figure 34 (Plate 4). The DO concentration has a major influence on the potential for denitrification to occur, as well as the oxidation state of nitrogen and phosphorus. When the DO concentration is less than 1 to 2 mg/L and organic carbon (or other electron donor) is in sufficient supply, bacteria convert nitrate to nitrogen gas (Buss et al., 2005; Bates and Spalding, 1998). Denitrification can also occur at microsites in soil and aquifer materials, where the bulk DO is higher than 2 mg/L, but is generally more significant where the bulk water is also low in DO (U.S. EPA, 2013).

DO concentrations were consistently above 2 mg/L in samples from wells AKG721 and AKG725 in the northwest part of the field. Most measurements in these wells were in the range of 6 to 10 mg/L, far above the level where denitrification occurs. In the other shallow monitoring wells, DO concentrations followed a seasonal pattern, with oxygen decreasing during the late summer, sometimes below 2 mg/L, probably due to elevated bacterial activity.

In the winter, shallow groundwater DO concentrations were rapidly replenished with oxygen-rich recharge from precipitation. The highest DO concentrations in most wells occurred in January and February following a water table rise of up to 7 feet within several weeks. Monitoring wells on the east side of the site followed this pattern most closely (e.g., AKG722).

DO concentrations in the deep well, AKG726, and the upgradient private wells were consistently at or close to 0.0 mg/L (anoxic). Occasional observations above 0.2 mg/L in AKG726 were not made using the standard sealed flow cell, because it was unavailable. Instead purge water was directed into the bottom of a 5-gallon bucket with the DO probe also at the bottom. The slightly higher values are probably an artifact of measuring in an open container.

Specific conductance and total dissolved solids (TDS)

Specific conductance and TDS results (Figures 35 and 36, Plate 4) followed similar patterns throughout the study. Both parameters generally increased in the fall-winter of 2004-2005, declined slightly, and peaked again in summer 2005. During the following 3 years, both parameters gradually declined until December 2008, when 3 of the shallow wells (AKG722, AKG723, and AKG725) showed substantial increases.

All conductivity and TDS results were below the secondary maximum contaminant levels (MCLs) for drinking water, 700 umhos/cm for conductivity and 500 mg/L for TDS (Chapter 246-290 WAC).

Chloride

Chloride concentrations are shown in Figure 37 (Plate 5) and ranged from 4.4 to 30.6 mg/L with the highest concentrations in 2004 and 2005. All results were below the secondary MCLs for drinking water of 250 mg/L (Chapter 246-290 WAC).

Patterns observed in chloride concentrations were similar to those observed for specific conductance and TDS. Concentrations of chloride were initially higher in the shallow well, AKG725, than in the nearby deep well, AKG726. Like most of the shallow wells in the study, chloride decreased at AKG725 for the first 3 years of the study until late 2008. Occasional increases in chloride at AKG725 corresponded with manure applications followed by heavy precipitation (see Figures 19 and 37, Appendix J). A particularly large increase in chloride occurred at AKG725 on August 16, 2005, when chloride reached 30 mg/L, indicating leachate reaching the water table even in the summer.

Chloride decreased slightly over time in the deep well, AKG726. Chloride concentrations in upgradient domestic water supply wells ranged from 16 to 18 mg/L.

Dissolved organic carbon (DOC)

DOC results are shown in Figure 38 (Plate 5). All organic carbon data collected before February 5, 2005 represent total organic carbon (no filtering). Samples collected on February 5, 2005 and afterward were filtered in the field and represent DOC except samples from AKG726. Samples from AKG726 were not filtered and represent total organic carbon, because the in-line filtering system was not equipped for the submersible pump needed for the deeper well.

AKG722 consistently had the highest DOC concentrations, with seasonal fluctuations that mimicked the water table elevations with about one month lag time (Figure 39). The maximum DOC observed was 9.6 mg/L at AKG722 on February 27, 2008. DOC in the other shallow wells sometimes fluctuated with the water table elevation but to a lesser extent than at AKG722.

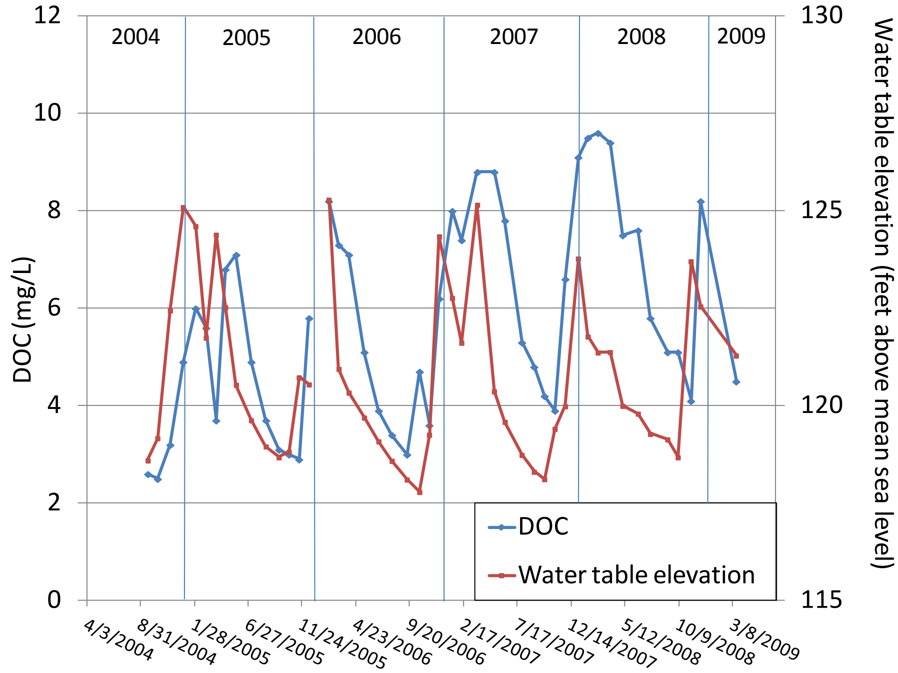

Figure 39. Dissolved organic carbon concentrations and water level elevations in monitoring well AKG722.

Nitrate

Nitrate results are shown in Figure 40 (Plate 5). Nitrogen was predominantly in the nitrate form in all wells except the deep, mostly anoxic well, AKG726. Concentrations of nitrate in shallow groundwater were well above 10 mg/L-N at the beginning of the study, except at AKG722. During the winter months of 2004 to 2005, nitrate concentrations increased in all wells to the highest levels observed during the study. The concentration at AKG722, 45 mg/L, was more than 4 times higher than the Washington State groundwater standard and federal MCL for drinking water.

During the second and third years of the study, 2006 to 2007, nitrate concentrations in the shallow wells decreased at about the same rate as declines in chloride concentration. Within the general decline, nitrate fluctuated somewhat. During the last few months of the study, December 2008 through March 2009, nitrate increased substantially in 4 out of 6 shallow wells, one reaching 20 mg/L-N (AKG725). Chloride followed a similar pattern.

In the anoxic deep well, AKG726, nitrate concentrations ranged from below the detection limit of 0.01 mg/L-N to 0.333 mg/L-N.

Concentrations of total persulfate nitrogen (TPN) were very similar to nitrate-N in shallow monitoring wells, indicating that little organic nitrogen was present.

Unfiltered nitrate concentrations in upgradient domestic water supply wells screened at 29 to 33 feet depth were 0.014 to 0.021 mg/L-N.

Ammonium

Results for ammonium-N are shown in Figure 41. Ammonium is typically attenuated in the soil, because positively charged ammonium ions ($NH4^+$) adhere to negatively charged soil particles (Buss, 2004). Because the pH was below 6 in the shallow monitoring wells and below 7 in the deep well, almost all of the ammonia is in the ammonium form. Therefore, for groundwater we will refer only to ammonium-nitrogen in this report.

Ammonium concentrations in shallow groundwater were below the detection limit of 0.01 mg/L on 37 out of 46 dates. The highest number of detections occurred on October 18 to 19, 2004, when the range of concentrations in 6 wells was 0.012 to 0.018 mg/L-N. Ammonium was the main nitrogen species found in samples from the deep well, AKG726, with concentrations ranging from less than 0.170 to 0.248 mg/L-N. Ammonium concentrations in upgradient domestic wells ranged from 0.194 to 0.255 mg/L-N.

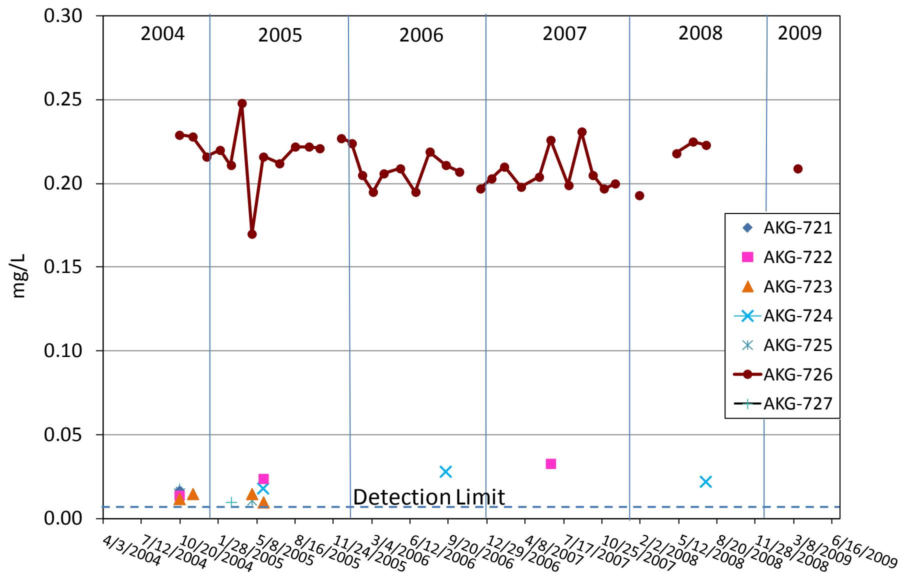

Figure 41. Ammonium-N results in monitoring wells.

Total persulfate nitrogen (TPN)

TPN concentrations were very similar to nitrate and are listed in Appendix R, Table R.1.

Total dissolved phosphorus (TDP)

TDP concentrations were monitored only from 2004 to 2006, because TDP was not the main focus of the study. Samples from AKG726 were not filtered and therefore represent total phosphorus (TP).

Results for phosphorus in groundwater are shown in Figure 42 (Plate 5). Shallow groundwater TDP ranged from 0.0005 to 0.0129 mg/L. Results for TP in the deeper anoxic groundwater were somewhat higher (0.113 to 0.264 mg/L) but are potentially biased high, because the samples were not filtered.

Discussion

The study results are discussed below from the perspective of nitrate movement, groundwater quality and aquifer characteristics. Manure, soil, and crop results are used to interpret groundwater results. For a detailed interpretation of manure, soil, and crop results, see VanWieringen and Harrison (2009).

Factors that influence nitrate conditions in groundwater and soil

The groundwater and soil nitrate conditions observed during the study were the result of a complex interaction of many environmental and farm management factors. These factors, and their influence on nitrate dynamics and fate, are discussed individually below.

Environmental factors

Hydrogeologic conditions

The Abbotsford-Sumas Aquifer, of which the Sumas-Blaine Aquifer (SBA) is a part, is not homogeneous. Grain size and depth to water vary spatially. These characteristics can play an important role in the vulnerability of different portions of the SBA to nitrogen leaching. Properties of the vadose zone and aquifer at the study site are discussed below and compared with information from other parts of the aquifer.

Grain size distribution

Movement of water and dissolved nitrate to the water table and along the groundwater flow path are affected by the texture of the substrate. Water penetrates more slowly through finer materials, which can lead to higher winter surface runoff and reduced nitrogen loading to the aquifer.

Finer soils over the central and western parts of the SBA tend to become saturated in the winter due to lower infiltration capacities, flat topography, and rapidly rising water tables. Recharge water percolating to the water table is enriched in oxygen. During the summer, when recharge is limited, oxygen can become depleted by bacterial consumption in finer soils, and denitrification is more likely. Slower velocity in fine-grained materials also allows more time for bacterial consumption of oxygen than in coarser materials.

Denitrification, which removes nitrate from the subsurface, is less likely in the coarser, well-drained soils and aquifer material in other parts of the SBA. There is more interconnected pore space in coarse-grained material, infiltration rates are faster, and replenishment of oxygen exceeds oxygen consumption by microorganisms. Paul and Zebarth (1997) found that denitrification accounted for only 17% of annual nitrogen loss from medium to coarse soils in south-coastal British Columbia following dairy manure application. The remaining 83% of soil nitrate was presumed to leach to groundwater.

The Natural Resources Conservation Service (NRCS, 2006) estimates nitrate loss of 15% in western Washington soils, where the depth to water is greater than 2 feet below the soil surface. Van Es et al. (2006) found that the nitrate concentration in drainage outflow beneath manured fields with loamy sand (coarse-grained) was on average twice what emerged from beneath loamy clay (fine-grained). de Ruijter et al. (2007) likewise found consistently higher groundwater nitrate concentrations in coarse-grained materials than in fine-grained materials at 34 farms studied in the Netherlands.

Figure 43 (Plate 1) compares the effective grain size results from well borings at the study site with locations shown on Figure 44 (Plate 1). Samples from 5-foot to 30-foot depths tend to grade coarser from west to east, with the Abbotsford samples having 10 times higher effective grain size values, at 16 and 26 feet, than the other sites. The Abbotsford samples indicate much coarser material in the northeastern part of the SBA than found to the west and southwest.

Particle size analyses from monitoring-well core samples indicated relatively fine-grained material at the screened depth, where sample water enters the well, in 4 out of 6 monitoring wells (sand with silt or clay) and coarser sand in 2 of the wells (AKG724 and AKG725). We saw evidence of denitrification in the wells with finer-grained material (except AKG721) as well as one well in coarser-grained material (AKG724). Core samples from shallower depths (2.5 feet) were extremely fine-grained in most samples (d_{10} less than 0.001 at 3 out of 4 bore holes).

These patterns suggest that nitrate losses due to denitrification are probably higher at the study site than in coarse-grained parts of the SBA, and especially higher than in very coarse-grained eastern parts of the aquifer. Rates of nitrate infiltration may also be lower at the study site than in coarser areas of the SBA.

Hydraulic conductivity and groundwater velocity

Hydraulic conductivity (K_H) estimates for the study site, 36 to 69 feet/day (average 53 feet/day), were lower than 84% of wells in the SBA analyzed by Cox and Kahle (1999). The lower K_H indicates slower flow than most locations where information is available in the aquifer. Lower groundwater velocities can result in a higher denitrification potential, likely making the study site somewhat less vulnerable to nitrate contamination by leaching than areas with higher K_H values.

After entering the aquifer, water and solutes at the study site move mainly horizontally in a southerly direction below the site. The average groundwater flow velocity at the site, 0.78 foot/day, is lower than the average value for the Abbotsford-Sumas Aquifer reported by Cox and Kahle (1999) of 2 feet/day. Lower velocity may lead to lower dispersion in the aquifer and slower decrease in nitrate concentration along the groundwater flow path.

Moisture content and preferential flow

Flow of water through the vadose zone to the water table occurs through both saturated and unsaturated flow. In unsaturated soils, as water content increases, the hydraulic conductivity and soil pore-water drainage generally increase (Wierenga, 1995). The change in hydraulic conductivity over the range of unsaturated conditions is less pronounced for fine-grained soils like those at the study site than for coarse soils in some locations overlying the SBA. In general, however, more and faster downward flow of water would be expected from the vadose zone to the water table with increasing soil moisture during the wet season (October through March).

Flow of water and dissolved constituents, such as nitrate, below the surface via cracks and openings as small as a few microns in diameter, can be rapid, inherently variable over a field, and difficult to track (Nimmo, 2013; Osnoy et al., 2005; Selker, 1999). Downward movement of percolating water prevents crop uptake or other processes that might otherwise remove nitrate or alter the chemistry of the water.

We assumed that most of the nitrate lost to leaching below the field reached the water table during the high-precipitation period (October through March). However, downward movement probably also occurred during the rest of the year to a lesser degree, especially following irrigation events.

Depth to water

Shallow groundwater depth observed during this study provides a short transport route for nitrate and dissolved constituents to groundwater unless the percolating water is redirected via tile drains to a surface water body. Tile drains are not present in the study field but are common in the low-lying area overlying the SBA. The winter water table at the study site was typically within 0 to 4 feet of the surface, intersecting the root zone of the crop and resulting in direct dissolution of nitrate into groundwater.

Air temperature

Crop growth and nitrogen uptake generally increase with warmer temperatures, leaving less excess nitrogen in the soil at the end of the growing season. VanWieringen and Harrison (2009) evaluated the influence of temperature on crop removal during the study using 3 methods that estimated growing degree units (GDUs). All 3 methods indicated that 2008 was significantly cooler than the other years and that most of the year-to-year variation in grass yield was due to temperature.

The Griffith and Thompson (1996) method used by VanWieringen and Harrison (2009) for estimating GDUs appears to best fit the study location and crop and is shown in Equation 8.

$$GDU = [(T_{Max} + T_{Min})/2] - 32 \qquad \textit{(Eq. 8)}$$

where:
GDU = Monthly growing degree units (°F)
T_{Max} = Maximum monthly temperature (°F)
T_{Min} = Minimum monthly temperature (°F)

The annual total GDUs for January through October are shown in Figure 45 (Plate 6). The highest year for thermal input to the grass crop, 2007, coincided with the highest crop nitrogen removal (Figure 46, Plate 6), one of the lowest annual nitrogen application rates (Figures 47, Plate 6), and groundwater nitrate values fluctuating around 10 mg/L-N (Figure 40, Plate 5). Likewise the year with the lowest annual GDU total, 2008, coincided with the lowest crop nitrogen removal. The lower crop removal and higher excess nitrogen in 2008 resulted in a return to shallow groundwater nitrate concentrations above 10 mg/L-N in most wells.

Influence of upgradient conditions on groundwater quality

Six out of 7 monitoring wells used for this study were constructed in the uppermost portion of the SBA. The open intervals of the wells intersected the water table, where the most recently recharged water was located. This allowed us to monitor groundwater quality responses to nitrogen cycling and loading in the near vicinity of each shallow well.

A shallow well specifically designed to represent the ambient groundwater quality at the upgradient boundary of the field was not installed for the study. Groundwater samples collected from the upgradient domestic wells are not considered representative of the *shallow* background condition at the far northern (upgradient) end of the study field, because of the position of their deeper open intervals 29 to 33 feet below ground surface (Figure 11).

Although the water quality of the shallow groundwater entering the upgradient boundary of the site was not directly monitored, it is unlikely that conditions beneath properties north of the study area had a significant influence on sample results during the study period. Our reasoning for this conclusion includes:

- The upgradient location, where a groundwater sample from a given depth first entered an aquifer, can be approximated using the following set of equations (Harter, 2013):

  $$D_{Recharge} = \frac{R}{n_e} \qquad (Eq. 9)$$

  where:
  $D_{Recharge}$ = the maximum depth of recharge movement into the aquifer (feet)
  R = the total amount of recharge during the period of interest (feet)
  n_e = the effective porosity of the aquifer sediments (dimensionless)

  then:

  $$GW_{Age} = \frac{R}{R_{Annual}} \qquad (Eq. 10)$$

  where:
  GW_{Age} = the maximum age of a groundwater sample of a given depth (year)
  R_{Annual} = the total annual recharge rate (feet/year)

  then:

  $$TD_{GW} = GW_{Age}\ x\ V_{GW} \qquad (Eq. 11)$$

  where:
  TD_{GW} = the approximate distance a groundwater sample collected from a given depth traveled in the aquifer from its point of entry as recharge (feet)
  V_{GW} = the average linear groundwater velocity (feet/year)

  An estimate of the maximum age of a groundwater sample collected from a well can be made by setting the $D_{Recharge}$ term equal to the maximum depth of the well's open interval below the water table and rearranging Equation 10 to solve for (*R*). Once determined, the maximum age of the sample (i.e., the age of the water drawn into the lowest point of the open interval) can then be integrated with the linear groundwater flow velocity at the site to approximate the greatest distance the sample traveled before arriving at the well.

The lowermost portions of the open intervals of the shallow wells at the site were typically no greater than 9 feet below the water table. Applying site-specific values to the equations above ($n_e = 0.25$; $V_{GW} = 285$ feet/year; $R_{Annual} = 2.25$ feet; $D_{Recharge} = 9$ feet), the approximate maximum travel distance (from the original point of entry as recharge) for groundwater samples collected from these wells is 285 feet. Water entering the well through the uppermost portions of the open interval is likely more recently recharged and has traveled a shorter distance through the aquifer.

Although this method greatly simplifies groundwater transport through an aquifer and into a well, the analysis suggests that samples collected from the shallow monitoring wells largely originated within the study field itself. Most of the monitoring wells are likely too shallow to capture any significant fraction of water originating upgradient of the study field. The mid-field and southernmost shallow wells (AKG722, AKG723, AKG724, and AKG725) were all located more than 500 feet downgradient from the upgradient edge of the field (Figure 48). This suggests that by the time groundwater travels from upgradient offsite sources to these wells, it has moved too deep into the aquifer to intersect the shallow well screens.

- Groundwater samples from the on-site well screened at the base of the aquifer (AKG726) are not considered representative of current management practices at the study site. Because the open interval for this well (28 to 38 feet) is so deep, samples from AKG726 probably represent water that entered the aquifer far upgradient of the study field (~1,200 feet, based on Equations 9-11). The effects of denitrification near the base of the aquifer appear to have removed most of the nitrate that was initially present in the groundwater when it first arrived at the water table.

- A 3.7-acre residence lies immediately upgradient (north) of the study site. The potential upper-range nitrogen input from the on-site sewage system at the residence was 36 lb nitrogen/year [9 lb nitrogen/person/year according to U.S. EPA (2002) times 4 residents living in the house full time] and potentially approximately 85 lb nitrogen/acre for lawn care on 2 acres (170 lb nitrogen total).

  The total estimated annual nitrogen loading to 3.7 acres would therefore be 206 lb nitrogen/ 3.7 acres or 56 lb total nitrogen/acre. This is 8 to 13% of the amount of total nitrogen applied to the study field between 2005 and 2008. This suggests that the water quality impact on the closest shallow monitoring well to this property (AKG721) was relatively insignificant in comparison to manure loading at the study site.

- The property farther upgradient of the residence was cultivated during the study and probably received manure at a rate similar to rates observed at the study field. However, the field lies 390 feet north of the upgradient edge of the study field, beyond the distance that samples from the shallow monitoring wells would likely be affected.

On the basis of the evidence above, the groundwater quality results presented in this report are assumed to largely represent groundwater quality responses to nitrogen loading and management occurring directly on the surface of the study field.

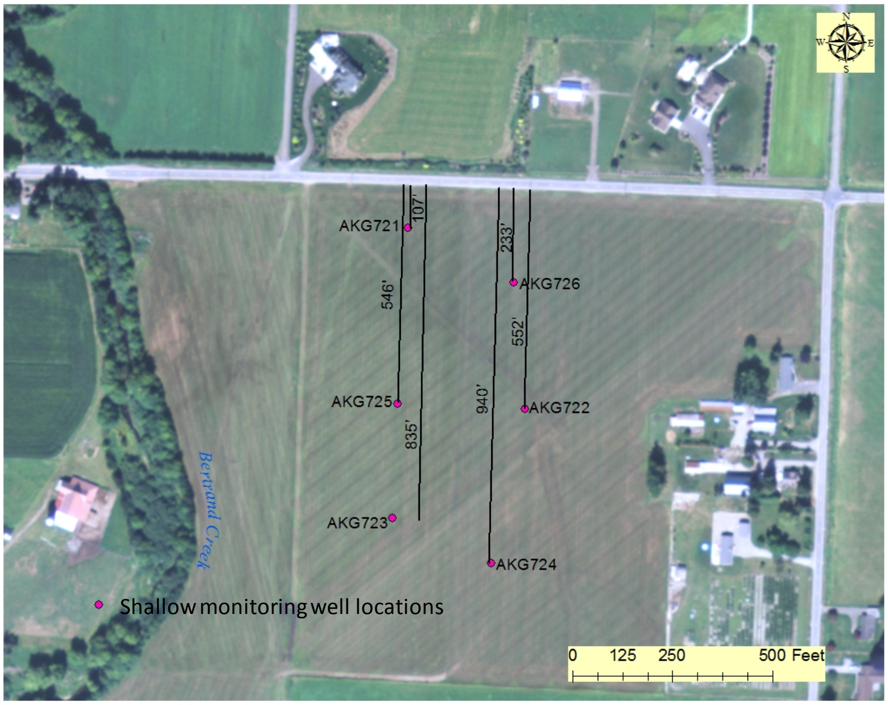

Figure 48. Distances from the edge of the study field to shallow monitoring wells.

Management factors

Rate of nitrogen application - external loading

The amount of nitrogen applied to the ground (from the combination of manure and inorganic fertilizer) has a significant effect on the amount of nitrogen available for leaching to groundwater. In general, the more nitrogen added to a field during the growing season in excess of the crop uptake and removal, the higher the amount of nitrate reaching the water table (Figure 49).

From 2004 through 2007, groundwater nitrate concentrations decreased steadily as a result of decreasing nitrogen load. Chloride, an indicator of nitrate loading, also decreased steadily until late 2008 (Figure 49). Both chloride and nitrate increased again in early 2008 as the nitrogen load increased. Such a rapid groundwater response to changes in nitrogen loading is due to the shallow depth to water and the high seasonal recharge rate.

The relationship between application rate and groundwater nitrate is not exact, because of the many other factors that play a role in nitrogen fate after manure is applied (e.g., availability of manure nitrogen for plant uptake, weather, crop performance, internal nitrate loading from mineralized organic matter). We compared the groundwater nitrate concentration for the typical time period when nitrate left over at the end of the growing season would have leached to, and been most evident in, shallow groundwater (November to December) to the annual amount of nitrogen applied to the field (manure and inorganic fertilizer) (Figure 48). The correlation was not significant ($r^2 = 0.178$).

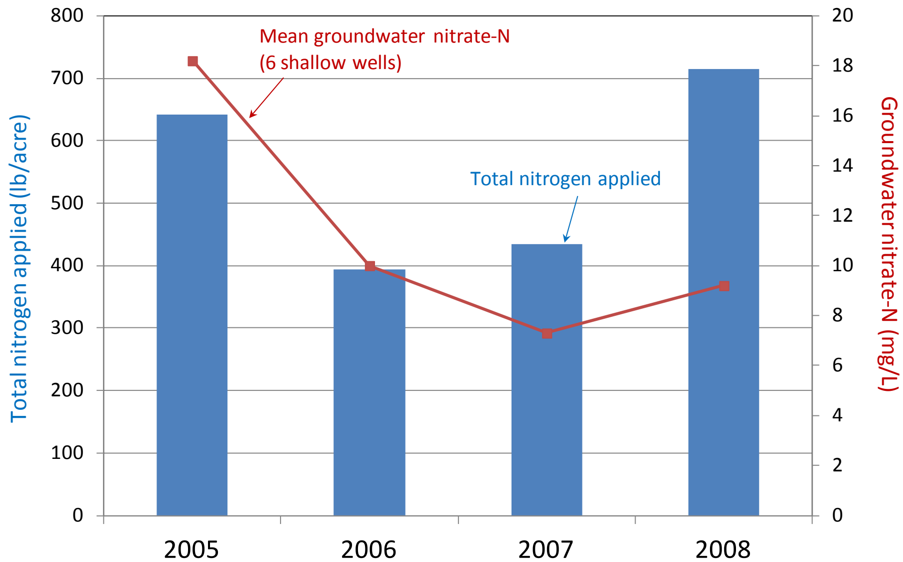

Figure 49. Total nitrogen mass applied (external loading) annually in manure and inorganic fertilizer (bars) and mean early winter (November and December) groundwater nitrate-N concentration in the 6 shallow wells.

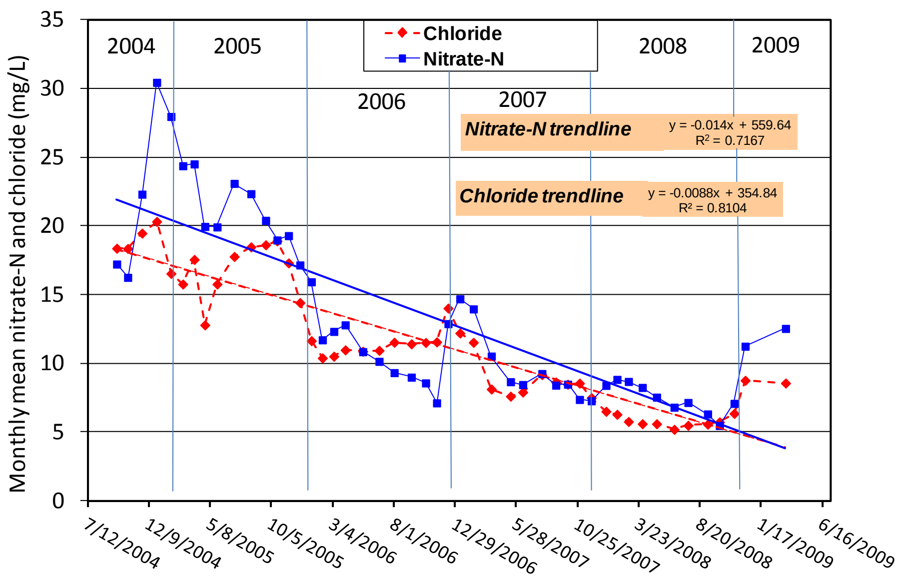

Figure 50. Mean shallow groundwater nitrate-N and chloride concentration trends over time.

The first year that we were able to record nitrogen loading, 2005, was notable for the high rate of external loading. Although unmeasured internal loading contributed additional nitrate to the field that year (due to enhanced mineralization triggered by tillage – discussed below), the high external loading in 2005 was probably the dominant factor affecting subsequent winter groundwater nitrate concentrations. This conclusion is supported by the similarity in groundwater chloride and nitrate trends during the study.

As shown in Figure 50, chloride, an indicator of nitrate loading, decreased steadily from 2004 through 2007. Like nitrate, chloride is a conservative element that does not adsorb to soil particles and is associated with manure application (Rodvang et al., 2004). Manure is the only major source of chloride in the area, and the mass of chloride in the soil that originates from manure would not be affected by tillage (Cogger, 2013). Both chloride and nitrate increased again in early 2008 as the external nitrogen load increased.

Rate of nitrogen mineralization – internal loading

In addition to nitrogen loading to the field from external sources, internal loading via mineralization of organic nitrogen in the soil to nitrate affects the amount of nitrogen available for leaching to groundwater (see *Nitrogen Cycle* section above). It is difficult to accurately quantify the amount and fate of nitrate produced by mineralization.

Tillage effects on mineralization

The highest groundwater and soil nitrate concentrations occurred during the fall and winter of 2004 to 2005. Although this was largely driven by a high rate of external loading, mineralization triggered by tillage of the field earlier in 2004 likely contributed to this condition. Tillage often leads to mineralization of accumulated organic nitrogen, resulting in higher than normal loading of nitrate to the underlying groundwater the following year (Rekha et al., 2011; Oenema et al., 2010; Gupta et al., 2004; Goulding et al., 2000; Whitmore et al., 1992).

Researchers have found conflicting results regarding the influence of tillage on nitrate leaching. Gupta et al. (2004) found higher nitrate leaching after tillage on manured fields than on non-manured fields. This was due to the gradual mineralization of organic matter from manure. Di and Cameron (2002) found the opposite result, higher leaching under no-till fields than under conventionally tilled fields.

Gupta et al. (2004) found higher percolation rates in no-till fields but similar nitrogen flux in manured and non-manured fields. The explanation offered for these differences is that a greater number of worm holes in no-till fields may provide more or better preferential flow paths than in tilled fields. The difference in impact between different cultivation methods may also be due to soil physical and biological factors (Rekha et al., 2011).

Plowing and replanting of the study field in spring 2004, after the field had been unplowed for several years, was at least partly responsible for a significant input of nitrate to the water table in late 2004 to early 2005. The highest monthly mean shallow groundwater nitrate concentration observed during the study, 30 mg/L-N, occurred on December 28, 2004 (Figure 50). This occurred 8 months after tillage, following a rise in the water table to within 1.3 to 4.1 feet of the ground surface (Figures 29, Plate 3). High fall recharge that caused the water table to rise also transported excess nitrate to the water table.

As mentioned previously, manure is the only major source of chloride at the study site and would not be affected by tillage (Cogger, 2013). Therefore, the significant decrease in both nitrate and chloride concentrations in groundwater from 2004 through most of 2008 indicates that release of nitrate from soil tillage was not the only reason for high nitrate concentrations in late 2004 and early 2005 (Figure 49).

Post growing season mineralization

The cumulative impact of soil nitrogen mineralization on groundwater nitrate leaching is difficult to quantify. Although the rate of mineralization is significantly influenced by temperature, it has been shown to be a significant year-round process (Zebarth, 2013; Zhao et al., 2010; Trindade et al., 2001).

Although a portion of newly mineralized nitrate may be taken up by the grass crop during the winter at the study site, rapid downward water movement probably leaches much of this internal load to the water table, 0 to 7 feet below the ground surface (Zebarth, 2013; Kuipers et al., 2012). Nitrate produced through non-growing season mineralization is generally not accounted for in growing-season mass balance analyses or in post-harvest fall soil nitrate sampling. Nitrate formed in soils through this on-going internal process is potentially available for leaching.

During the winters of 2004-2005 and 2007-2008, nitrate results from several shallow wells indicated a nitrate source other than recently applied manure. Chloride concentrations decreased or remained stable between November and February, while nitrate concentrations increased in 3 wells in 2004-2005 and 4 wells in 2007-2008 (Plate 7).

These nitrate increases in shallow groundwater appear to be from an on-site winter source, because nitrate and chloride concentrations typically track closely to one another absent additional loading. Nitrate and chloride additions from growing-season manure applications had most likely already reached the water table. Therefore, these winter nitrate increases could potentially be from soil nitrate that mineralized after the fall. Another possible source for the winter 2004-2005 groundwater nitrate increase could be lingering effects from nitrate mobilized by tillage, which also would not be linked with chloride.

Crop removal

The mean annual mass of nitrogen removed in the crop was 430 lb/acre (standard deviation = 27) (Figure 46, Plate 6). Consistent levels of nitrogen removal by the crop over 4 years indicate that the large range of nitrogen application rates (393-715 lb/acre) did not affect the crop removal of nitrogen (Figure 47, Plate 6).

Timing of manure applications

Timing of manure application had an effect on nitrate loss to groundwater. Manure applied just prior to major precipitation events at rates that resulted in excess soil nitrate were often followed first by higher soil nitrate values and then by higher shallow groundwater nitrate concentrations.

However, when manure was applied during relatively dry periods and in amounts that the crop could take up, no subsequent increase in soil or shallow groundwater nitrate was observed.

Precipitation at the end of the growing season typically triggers mineralization of soil and manure organic nitrogen sources even without manure application. Examples of manure application timing effects in spring and fall are described below.

Spring applications

Spring application of manure (February through April) can cause spikes in underlying groundwater nitrate concentrations if a large amount of precipitation occurs after the application. The timing and amount of spring manure application over the Sumas-Blaine Aquifer require extreme caution from a groundwater protection standpoint, because recharge is still occurring. Applying manure before the crop can efficiently take up nitrogen in the earliest days of the growing season risks leaching substantial nitrate to groundwater.

In the spring of 2005, although conditions were dry for several days before and after 2 spring manure applications, the total amount of nitrogen applied, 272 lb/acre, combined with nitrate mineralizing from soil organic matter, appears to have exceeded the new grass crop's uptake potential. Groundwater nitrate concentrations increased in all shallow monitoring wells by 4 to 13 mg/L-N during the following 3 months (Plate 8).

Wet weather in March and April 2005 (7 inches of rain in the 2 months following the February 18 application) contributed to downward water and nitrate movement as indicated by the 2.5-foot rise in the water table to within 2.4 feet of the ground surface at well AKG722 (Figure 28, Plate 3). When the water table is close to the surface, most of the nitrate in the soil is probably lost to groundwater (Plate 8).

In the spring of 2007, although the amount of manure nitrogen applied (240 lb/acre) was similar to the amount applied in spring 2005, a smaller and less immediate increase in groundwater nitrate was observed at 3 wells (AKG721, AKG723, and AKG727) (Plate 8). Nitrate concentrations in these wells increased by 2 to 5 mg/L following the 2007 spring manure applications. Precipitation during the month following the initial application, 8 inches, was similar to that in 2005, although evapotranspiration later in the year would have been higher. The smaller increase in groundwater nitrate following the spring 2007 manure applications compared to 2005 may be due to higher evapotranspiration and plant uptake as well as other factors related to the one-month later start date (March 14) in 2007.

Groundwater nitrate increases were less widespread following spring manure applications in 2006 and 2008 (Plate 8). In 2006, less nitrate reached the groundwater due to a combination of a smaller load than in 2005 and 2007 (171 lb/acre nitrogen), application later in the season (April 27), and essentially no recharge the month following application. Lower recharge in spring 2008 than in other years apparently prevented or delayed leaching of nitrate from the relatively heavy spring application of 364 lb/acre nitrogen in 2 episodes (March 10 and April 20).

Wet soil conditions in the spring can delay initial manure application as well as crop harvests throughout the growing season. In 2006, wet weather delayed the first grass cutting until April 2. The first manure application for the year therefore occurred 6 to 10 weeks later than in previous years. The time available for the repeated process of manure application, crop uptake, and crop removal was thereby shortened by several weeks. This led to a late final manure application in 2006 (October 5) followed by an increase in groundwater nitrate.

Fall applications

In the coastal Pacific Northwest, application of manure in the fall, even on perennial crops, presents a high risk of nitrate leaching (Kowalenko, 1987; Zebarth et al., 1998). In the fall, crop growth slows, precipitation increases, and leaching of nitrate is all but assured.

The consequences of applying manure too late in the fall are demonstrated in Plate 9. For example, a late final fall manure application in October 2006 resulted in an average increase in nitrate concentration of 6 mg/L-N in the shallow winter groundwater (maximum increase of 16 mg/L-N). In 2007, however, when the last application occurred earlier in the season (September), no significant increases in winter groundwater nitrate concentrations were observed. In fact, nitrate decreased by an average of 2.4 mg/L in 5 out of 6 shallow wells.

Most of the variables for nutrient uptake and application were similar in 2006 and 2007. The nitrogen application rate, temperature (growing degree days as defined in Equation 8), and crop removal were similar, although weather and crop removal were slightly better in 2007 (Figures 45 and 46 in Plate 6 and Figure 49). This suggests that the key difference between these 2 years was timing of the final manure application.

2006—Effects on groundwater from late fall manure application

Although the amount of nitrogen applied in the fall of 2006 was the second lowest fall manure application during the study (90 lb/acre), application late in the season on October 5 and pre-existing high soil nitrate levels led to significant effects on shallow groundwater nitrate concentrations.

By November 8, 2006, the soil nitrate concentration increased from 31 mg/kg (96 lb/acre) on October 1 to 60 mg/kg (186 lb/acre), 4 times the recommended level for nutrient balance and at a time well after the typical growing season (Figure 51, Plate 6). A 4.7-inch rain event in early November provided the recharge necessary to transport the nitrate downward. By November 15, 2006, the soil nitrate concentration at 1-foot depth decreased to 15 mg/kg (47 lb/acre), indicating significant leaching of nitrate from the root zone.

As the nitrate mass infiltrated downward in the fall of 2006, the water table rose 6.7 feet (Figure 28, Plate 3). Between October 18 and December 12, 2006, the mean shallow groundwater nitrate concentration increased by over 4 mg/L to 12.9 mg/L-N (Figure 51, Plate 6). Individual well nitrate concentrations increased by up to 16 mg/L-N (Plate 9). Simultaneous increases in chloride concentrations in individual wells indicate manure as the likely source of the nitrate increase (Plate 9).

Prolonged precipitation in the fall can also prevent crop harvest. In 2005, 102 lb/acre of nitrogen in the grass crop were not removed from the field due to wet weather. In such cases, the full benefit of crop uptake is not realized, because some of the crop nitrogen that would have been removed decays and may become available for leaching to groundwater.

2007—Effects on groundwater from early fall manure application

Early application of manure (September 7) and the lightest fall application during the study, 77 lb/acre, apparently allowed for crop uptake of nutrients before temperatures decreased substantially and before the onset of heavy rain in 2007. As a result, groundwater nitrate concentrations did not spike in the fall of 2007 as they did in 2006 (Plate 9). Nitrate concentrations in 4 out of 6 wells remained below 10 mg/L-N through February 2008.

These examples indicate that timing of manure applications during periods at the margins of the major growing season (fall and late winter) pose a high risk of nitrate leaching due to the combination of unpredictable influencing factors. Precipitation, especially heavy rain, during these times can transport mineralized nitrate (from current and past manure applications) below the root zone before crop uptake can occur.

Conditions during the late winter/early spring are particularly conducive to rapid leaching of available nitrate to the water table. Ammonium from manure applied during this high-recharge time eventually nitrifies to nitrate, while at the same time organic nitrogen remaining in the soil begins to mineralize to nitrate. Nitrate from both sources (winter/early spring manure application and mineralized organic matter) is susceptible to leaching before the grass crop can take up the bulk of the load (Trindade et al., 2001; Zhao et al., 2010; Zebarth, 2013).

Soil moisture

During the growing season, insufficient soil moisture inhibits crop growth. When crop growth is inhibited, less nitrogen is taken up by the crop. This leads to an excess of nitrogen in the soil that can leach to groundwater. In the summers of 2005 and 2006, soil moisture declined to levels that restrict grass growth and bacterial mineralization of ammonium and organic nitrogen (i.e., less than 20%). During this time, the grass crop probably went dormant, resulting in lower nitrogen uptake, lower yield, and more excess nitrogen than if more moisture had been available (VanWieringen and Harrison, 2009) (Figure 17, Plate 2).

In 2007, the dairy producer began irrigating in July, one month earlier than previously, to prevent crop dormancy and improve crop uptake of nitrogen. As a result, soil nitrate concentrations were lower and the crop uptake and removal were higher in the fall of 2007 than in 2006 (Figures 25 and 26, Plate 2).

While additional moisture in the summer of 2007 apparently resulted in higher crop uptake of nitrogen, too much irrigation water during the growing season can move nitrate below the root zone even when the soil is not saturated. This prevents possible crop uptake and contributes to higher groundwater nitrate concentrations. This may have occurred in July and/or August 2008, when the growing season was cooler and wetter than normal.

Denitrification

The dissolved oxygen (DO) condition in soil and groundwater can have a significant effect on groundwater nitrate concentration. When the groundwater DO is below 2 mg/L, denitrification can remove nitrate and lead to lower nitrate concentrations (Van Es et al., 2006; Rodvang et al., 2004). DO in groundwater varied spatially and over time in the shallow monitoring wells, indicating variability in the rate and timing of denitrification in groundwater.

Denitrification probably caused routine, periodic nitrate loss in groundwater in 4 of the 6 shallow monitoring wells, when the DO was 2 mg/L and lower (AKG722, AKG723, AKG724, and AKG-726 – Plate 8). Denitrification probably also occurred above the water table in these areas before leachate reached the water table, which is typical of wet, fine-textured soils (Coyne, 2008; Paul et al., 1997; Murray et al., 1995). Denitrification above and below the water table therefore likely muted the effect of excess nitrogen infiltrating to groundwater, especially from the wells with low DO.

Nitrate: chloride ratio as an indicator of denitrification

As described earlier, chloride is largely non-reactive in the subsurface. Changes in chloride concentrations in groundwater are therefore assumed to be due to either dilution or a change in chloride loading. These characteristics make chloride useful for evaluating nitrate changes in groundwater (McCallum et al., 2008; Rodvang et al., 2004).

An additional tool for evaluating whether denitrification is a major factor in controlling groundwater nitrate concentrations is to compare ratios of nitrate-N to chloride (NO_3-N:Cl) in groundwater (McCallum et al., 2008). Variations in the ratio of nitrate-N to chloride can indicate loss of nitrate due to denitrification where DO is sufficiently depleted.

Nitrate and chloride consistently followed similar patterns in the 2 high-DO wells (AKG721 and AKG725; Figure 52, Plate 6). Initially both wells had a higher concentration of nitrate than chloride. This was probably due to increased mineralization of soil organic matter related to tilling the field. After the first year, however, nitrate and chloride concentrations corresponded more closely.

During months with little or no recharge, and when the water table is too far below the root zone for plant uptake (June through September), relative differences in the proportion of nitrate and chloride concentrations (NO_3-N:Cl) are probably due to denitrification rather than to crop uptake or management activities.

Nitrate and chloride concentrations generally tracked closely when DO was above the 2 mg/L threshold for denitrification in the 4 wells with at least occasionally low DO (Figure 53, Plate 6). However, when DO fell below 2 mg/L, nitrate and chloride concentrations diverged, indicating loss of nitrate to denitrification. For example, in the fall of 2006, when DO in AKG722, AKG723, AKG724, and AKG727 was below 2 mg/L (Plate 9), chloride concentrations remained at the same level or increased. During the same time period, nitrate concentrations dropped by 4-10 mg/L-N (Figure 53, Plate 6).

The mean NO_3-N:Cl ratios for the 2 DO conditions were:

- High-DO wells (always greater than 2 mg/L): 1.39 (SD[9]=0.26, n=96)
- Low-DO wells (less than 2 mg/L at times): 1.05 (SD=0.44, n=188)

[9] Standard deviation

The NO_3-N:Cl ratio was 32% higher in the high-DO wells than in the low-DO wells throughout the study and was relatively stable. The higher variation, as indicated by the higher standard deviation, in the seasonally low-DO wells is consistent with the fluctuation in DO concentrations above and below the denitrification threshold.

The nitrate losses indicated by the NO_3-N:Cl ratio suggest an average 28% reduction in nitrate in the low-DO wells compared to the high-DO wells. For example, if 2 groundwater samples from the site each contained 10 mg/L chloride, the sample from a high-DO well would have a nitrate concentration of 13.9 mg/L-N on average. The sample from a low-DO well with 10 mg/L would have a nitrate concentration of 10.1 mg/L-N on average.

McCallum et al. (2008) found a similar distinction between NO_3-N:Cl ratios in groundwater beneath manured fields where denitrification was occurring and manured fields where denitrification was not occurring.

Summary of major influences on groundwater nitrate conditions at the study site

The management and environmental factors discussed above interact and influence both the amount of nitrate that leaches to the water table and the resulting groundwater nitrate concentrations in a complex manner. The data collected during this study suggest that groundwater nitrate conditions at the study site were particularly sensitive to:

- Manure applications that significantly exceeded crop demand.
- Manure applications that occurred outside of the typical growing season, particularly during periods of high recharge.
- Initial tillage event.
- Denitrification capacity of the aquifer.

Annual nitrogen balance and residual estimation – correlation to groundwater quality

A primary goal of Dairy Nutrient Management Plans (DNMPs) is to closely balance nutrient application and plant uptake to minimize residual or excess nitrogen remaining at the end of the growing season. Residual nitrogen in nitrate form can be transported to the water table, where it can impact groundwater quality (Harter and Menke, 2005; Zebarth et al., 1998).

Two field-based methods for evaluating the end-of-growing-season nitrogen residual are:

1. Indirect estimation by mass balance analysis.
2. Direct estimation by end-of-season soil sampling.

Below we discuss annual nitrogen residuals predicted by these 2 approaches and compare the estimates to observed groundwater conditions beneath the study field.

1. Estimate of annual nitrogen residual by mass balance analysis

The various nitrogen inputs and outputs measured during the study can be incorporated into a mass balance analysis to estimate the residual amount of nitrogen mass present at the end of each growing season using Equation 12. The method assumes that measurements of nitrogen inputs and outputs are complete and that there are no unidentified gains, losses or storage in the system. In cases where an equation variable was not directly measured in the field, an estimate for the value was developed using standard industry assumptions. This method also assumes that all excess nitrogen is converted to the nitrate form and leaches to groundwater at the end of the growing season.

$$N_{Excess} = N_{Input} - N_{Output} \qquad (Eq.\ 12)$$

where:
N_{Excess} = Excess nitrogen mass left in the soil column at the end of the growing season (lbs/acre)
N_{Input} = Total nitrogen mass input during the growing season (lbs/acre)
N_{Output} = Total nitrogen mass output during the growing season (lbs/acre)

$$N_{Input} = PAN + F + A + I \qquad (Eq.\ 13)$$

where:
PAN = Plant-available nitrogen from manure and soil organic matter (lbs/acre—calculated based on measured manure N)
F = Nitrogen applied as inorganic fertilizer (lbs/acre, estimated)
A = Nitrogen from atmospheric deposition (lbs/acre, estimated)
I = Nitrogen applied in irrigation water (lbs/acre, measured and estimated)

and

$$N_{Output} = Y + V + D + L \qquad (Eq.\ 14)$$

where:
Y = Nitrogen removed in crop (lbs/acre, measured)
V = Nitrogen lost to volatilization during application (lbs/acre, estimated)
D = Nitrogen lost to denitrification (lbs/acre) (term ignored for this analysis)
L = Nitrogen lost to leaching during the growing season (lb/acre) (term ignored for this analysis)

Nitrogen losses due to denitrification in the unsaturated zone (above the water table) were considered negligible during the growing season (Sullivan, 2008); therefore, the variable D in Equation 14 was ignored. Leaching losses were also assumed to be negligible for the analysis, because recharge is very low during the growing season. Therefore, variable L in Equation 14 was also ignored.

Three methods, described below, were used to estimate plant-available nitrogen (PAN), the main component of N_{Input} in Equation 12.

Method 1-PAN

Method 1 for estimating PAN is based on Sullivan's (2008) method, as shown in Equation 15.

$$PAN = 0.85M_{NH4} + 0.4(M_{OrgN}) + 0.15(OrgN_{Yr\,1}) + 0.07(OrgN_{Yr\,2}) + 0.03(OrgN_{Yr\,3}) + 0.02(OrgN_{Yr\,4-9\,Sum}) \qquad (Eq.\ 15)$$

where:
M_{NH4} = ammonium-N content of the manure applied during the current year (lbs/acre)
M_{OrgN} = organic N content of the manure applied during the current year (lbs/acre)
$OrgN_{Yr\,1}$ = organic N content of the manure applied the year before the current year (lbs/acre)
$OrgN_{Yr\,2}$ = organic N content of the manure applied 2 years before the current year (lbs/acre)
$OrgN_{Yr\,3}$ = organic N content of the manure applied 3 years before the current year (lbs/acre)
$OrgN_{Yr\,4\text{-}9\,Sum}$ = organic N content of the total amount of manure applied during the period 4 to 9 years before the current year (lbs/acre)

Equation 15 assumes that during the current year, 85% of the ammonium-nitrogen and 40% of the organic nitrogen in the manure is available to plants when applied using surface banding. An increasingly smaller portion of the organic nitrogen is available from previous applications up to year 4 before the current year, when the portion remains constant at 2% through year 9.

The mean annual organic nitrogen in manure during the study, 275 lb/acre/year, was used to estimate the nitrogen mineralized from organic matter for subsequent years (Appendix J, Table J.2).

Variable (*V*) in Equation 14 is ignored in the mass balance analysis using Method 1, because loss of ammonia to volatilization is accounted for in the method. The Method 1 PAN estimates developed for each study year are shown in Table 9.

Table 9. PAN estimates (lb/acre) using 3 methods.

PAN Method	2005	2006	2007	2008	Average
1	517	310	321	530	420
2	565	359	369	578	468
3	644	363	386	715	527

Equation 15 assumes that manure applied during the current year is in the form of a thin slurry (1 to 5% dry matter) and is applied using surface banding. Liquid manure applied during the study was 1 to 5% dry matter on 16 out of 17 dates and 7% on one date.

Method 2-PAN

Method 2 for estimating PAN uses a modification of the Sullivan (2008) procedure to account for the possible effects of relatively high organic matter measured in the study site soils (average 7.5%) (Zebarth, 2013; Sullivan, 2013).

Method 2 estimates the contribution of PAN from previous years' applications using either Equation 16 or 17:

When the organic matter in the soil is less than 5% (NRCS, 2006):

$$PAN = 0.85M_{NH4} + 0.4M_{OrgN} + 20(OM) \qquad Eq. 16$$

or, when the organic matter in the soil is equal to or greater than 5% (NRCS, 2006):

$$PAN = 0.85M_{NH4} + 0.4M_{OrgN} + 150 \qquad Eq. 17$$

where:
OM = average organic matter of the soil (%)

Method 2 also takes into account loss of ammonia to volatilization; therefore, variable (*V*) in Equation 14 is ignored for the mass balance analysis for this scenario. PAN estimates for Method 2 are shown in Table 9.

Method 3-PAN

Method 3 for estimating PAN uses the total nitrogen applied in manure for the year (Equation 18). Although not all of the total nitrogen applied is actually available that year, this method assumes that the mass of nitrogen that is not plant-available is balanced out with the amount that is mineralized from soil organic matter. Method 3 estimates for PAN are shown in Table 9. In contrast to PAN estimation Methods 1 and 2, the Method 3 PAN values do not account for ammonia volatilization; therefore, a separate value for variable (*V*) in Equation 14 is included in the mass balance analysis. (See *Other elements of the nitrogen balance* below.)

$$PAN = Total\ N_{Manure} \qquad Eq. 18$$

where:
$Total\ N_{Manure}$ = Total mass of nitrogen applied in manure for the year

Comparison of 3 methods for estimating nitrogen inputs

The 4-year mean for the 3 methods for estimating nitrogen inputs are shown in Figure 54. Methods 1 and 2 produced roughly the same mean value, 418 to 420 lb/acre, while Method 3 was 25% higher. Standard deviations for the 3 input methods were 24 to 34% of the means (or 98 to 179 lb/acre).

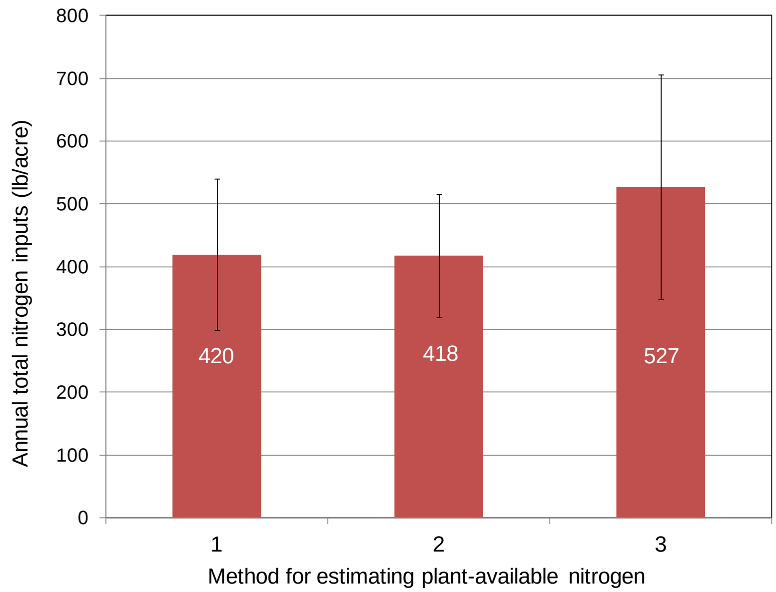

Figure 54. Mean annual nitrogen inputs estimated using PAN Methods 1 through 3. *Error bars indicate +/- one standard error. Numbers on the bars are the mean values.*

There is considerable uncertainty associated with each method for calculating the PAN component of N_{Input}. The most difficult PAN value to quantify is the amount of nitrogen mineralized from the complex mixture of organic compounds in manure and soil (University of California, 2007). While it is well documented that organic forms of nitrogen mineralize in a half-life decay pattern, with slower breakdown of more complex compounds over time, many factors affect the rate of decay. The estimates for nitrogen available from soil organic matter used in this analysis probably have a large margin of error.

The simplified manner in which Method 1 was applied (i.e., using a single value to represent the organic nitrogen content across multiple years) does not capture the variability in manure composition and loading within each year or between years. The value used in Method 1 for estimating PAN from soil organic matter, the mean annual organic nitrogen mass (275 lb/acre), may not represent the actual variation in organic nitrogen content in the manure applied in past years. The largest component of PAN in Method 1 (67 to 81%), however, comes from the current year, making past contributions less influential.

Method 2 provides an even less reliable estimate of the soil organic nitrogen contribution to the PAN pool than Method 1. Significant uncertainty exists regarding whether soil organic matter is a reliable indicator of PAN, because the organic matter in soils with the highest organic matter content is older and tends to decompose more slowly than organic matter in soils with lower percentages (Sullivan, 2013). Soils with lower organic matter content are typically of more recent origin, and this organic matter tends to mineralize more readily. NRCS (2006) presents a wide range for nitrogen mineralization in soils with 5% or greater organic carbon (50 to 200 lb/acre).

The soil organic nitrogen contribution to PAN in Method 3 is based on agricultural experience, but we were not able to confirm how well results for this method compare with field conditions.

Average PAN input using this method was 107 and 59 lb/acre higher than Methods 1 and 2 respectively (Table 9).

Other elements of the nitrogen balance

In addition to PAN, other elements measured during the study include nitrogen inputs from inorganic fertilizer and irrigation water (Equation 13) and nitrogen outputs from crop removal (Equation 14). Components that were not measured during the study [atmospheric deposition and volatilization (for Method 3 only)] were estimated based on the following assumptions:

- Annual atmospheric input of nitrogen was assumed to be similar to that reported at nearby Abbotsford, British Columbia, 8 lb/acre/year (Kuipers et al., 2012).
- For the Method-3 PAN scenario, 15 % of the annual PAN was assumed to be lost to ammonia volatilization. This assumes that subsurface deposition is equivalent to surface banding (partial incorporation) (Sullivan, 2008).

The estimated values for atmospheric inputs and ammonia volatilization for each year are presented in Table 10. The contribution of nitrate to the soil and groundwater from tilling the field in April 2004, due to enhanced mineralization of soil organic matter, was probably substantial (Oenema et al., 2010; Goulding et al., 2000; Whitmore et al., 1992).

Mass balance results – annual estimates of residual nitrate

Table 10 summarizes the annual N_{Input} and N_{Output} estimates for 2005 to 2008 using the 3 PAN estimation methods described above. The table also presents the N_{Excess} estimate for each year calculated using Equation 12. A mass balance analysis was not conducted for 2004, because detailed information on manure nitrogen inputs was not available.

The annual N_{Excess} estimates ranged from -79 to 146 lb/acre. The 4-year average mass balance residuals were 19 lb/acre for Method 1, 67 lb/acre for Method 2, and 79 lb/acre for Method 3.

The estimated annual nitrogen inputs and outputs are shown graphically in Figures 55 (inputs only) and 56, using the Method 1-PAN values. Because the Method 1-PAN estimation approach is based on the most up-to-date regional understanding of nitrogen dynamics in manured fields, subsequent analyses in this report will use only results of this method. Our qualitative judgment of the level of confidence in each input category is shown in the Figure 54 legend.

The mass balance analysis indicates:

- Approximately one-half of the nitrogen input comes from ammonium directly applied to the crop.
- Internal loading by mineralization of organic nitrogen contributes nearly as much as direct application.
- Other sources such as irrigation and atmospheric inputs represent only a small fraction of the overall nitrogen load.
- Crop removal is the dominant process for nitrogen output during the growing season.
- The Method 1 mass balance analysis indicates that substantial nitrogen deficits occurred during the 2006 and 2007 study years.

Table 10. Mass balance calculation of annual nitrogen (N) excess at the end of the growing season (lb/acre).

Method 1 - PAN	2005[1]	2006	2007	2008	Average
INPUTS (N_{Input})					
PAN from Sullivan (2008) (M)	517	310	321	530	
Inorganic fertilizer (F)	0	31	48	0	
Irrigation water (I)	0.8	1.8	1.7	1.4	
Atmospheric input (A)[2]	8	8	8	8	
N_{Input} totals	526	351	379	539	
OUTPUTS (N_{Output})					
Crop N removed (Y)	439	430	457	393	
N_{Output} totals	439	430	457	393	
N_{Excess} = (N_{Input} - N_{Output})	**87**	**-79**	**-78**	**146**	**19**
Method 2 - PAN	**2005[1]**	**2006**	**2007**	**2008**	
INPUTS (N_{Input})					
PAN from Sullivan (2008) for current year + 150 lb/acre SOM[3] (M)	565	359	369	578	
Inorganic fertilizer (F)	0	31	48	0	
Irrigation water (I)	0.8	1.8	1.7	1.4	
Atmospheric input (A)[2]	8	8	8	8	
N_{Input} totals	574	400	427	587	
OUTPUTS (N_{Output})					
Crop N removed (Y)	439	430	457	393	
N_{Output} totals	439	430	457	393	
N_{Excess} = (N_{Input} - N_{Output})	**135**	**-30**	**-30**	**194**	**67**
Method 3 - PAN	**2005[1]**	**2006**	**2007**	**2008**	
INPUTS (N_{Input})					
Manure total nitrogen applied (M)	644	363	386	715	
Inorganic fertilizer (F)	0	31	48	0	
Irrigation water (I)	0.8	1.8	1.7	1.4	
Atmospheric input (A)[2]	8	8	8	8	
N_{Input} totals	653	404	444	724	
OUTPUTS (N_{Output})					
Crop N removed (Y)	439	430	457	393	
Ammonia volatilized (15% of Ammonium-N applied) (V)	62	31	33	64	
N_{Output} totals	501	461	490	457	
N_{Excess} = (N_{Input} - N_{Output})	**152**	**-57**	**-46**	**267**	**79**

[1] Last grass cutting, 102 lb/acre, was not removed from the field but is included in crop nitrogen removal.
[2] From Kuipers (2012). [3] Soil organic matter.

Despite substantial estimated nitrogen deficits in 2006 and 2007, annual crop nitrogen removed from the field was roughly the same each year (Figure 46, Plate 6). The highest yield, 457 lb/acre N, was obtained in 2007, when the Method 1-PAN indicated the highest deficit, -79 lb/acre. The lowest yield observed, 393 lb/acre N in 2008, followed the highest annual Method 1-PAN mass balance excess (146 lb/acre). This indicates that plant nitrogen uptake was probably not limited by nitrogen application and suggests that more nitrogen was available for both plant uptake and leaching than indicated by the mass balance analysis.

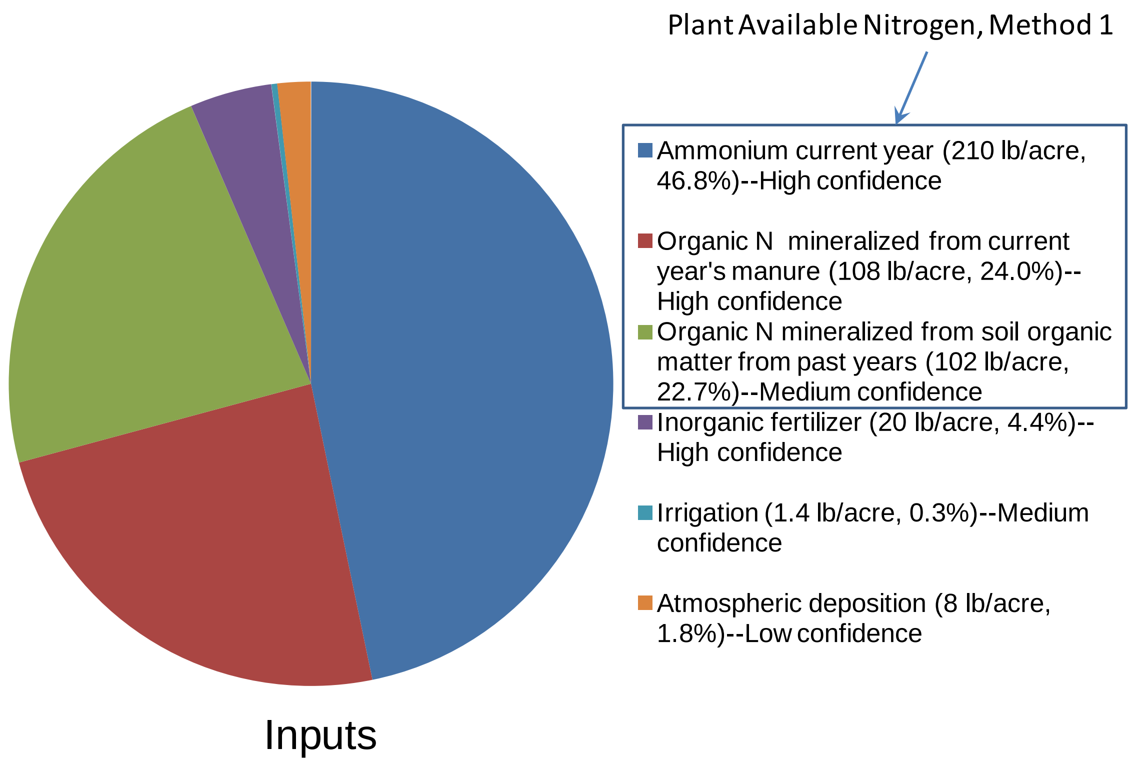

Figure 55. Average annual nitrogen inputs to the soil column during the study in lb/acre using Method 1 and our relative degree of confidence in the numbers.

Data are shown in Appendix S.

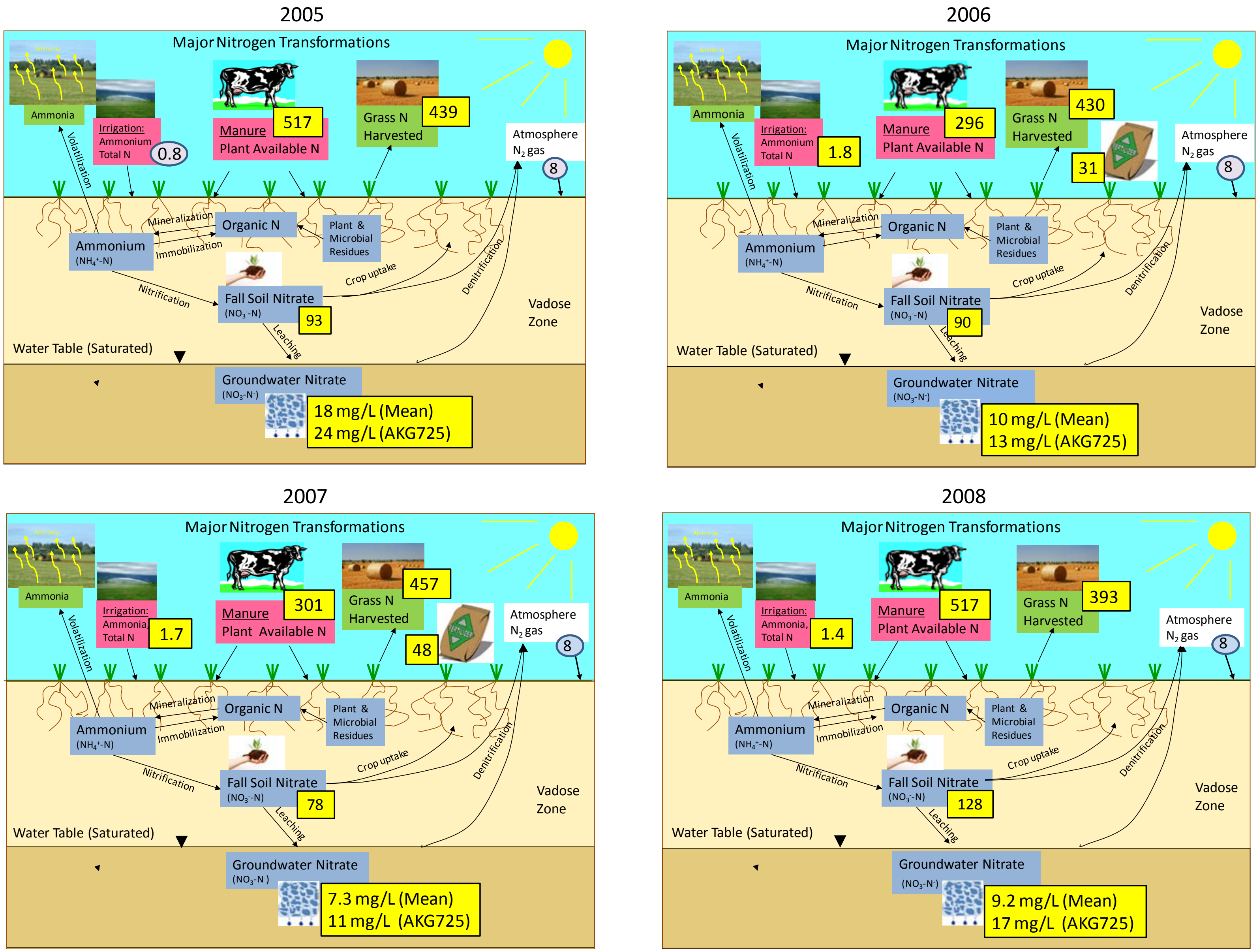

Figure 56. Annual nitrogen inputs, outputs, and soil nitrate residual.

See next page for explanation.

Explanation for Figure 56: Inputs, outputs, and soil nitrate are in lb/acre. Values in yellow boxes were measured. Values in blue ovals are estimates. Groundwater nitrate-N concentrations (Mean) represent mean winter (November to December) values for the 6 shallow monitoring wells are in mg/L; (AKG725) represents mean winter values for well AKG725. Plant Available N (shown in the Manure box) is from Method 1 in Table 9. Atmospheric N input is from Kuipers et al. (2012). Fall soil nitrate residual is the maximum of weekly values for September 1 through October 31. Denitrification outputs from soil are assumed to be negligible during the growing season.

Color code: green=outputs, pink=inputs, white=inputs and outputs, blue= resulting effects on soil and groundwater.

Sources of uncertainty in the N_{Excess} results

Most of the nitrogen inputs during the growing season were measured with high precision and accuracy and are based on direct field monitoring results (Figure 55). Outputs in the form of crop removal were also measured precisely and accurately. However, the methods for estimating a number of the other components of the mass balance have important limitations and therefore introduce a degree of uncertainty in the residual estimates. The major sources of uncertainty in the mass balance analysis include:

- The 2005 N_{Excess} estimates are biased low, because the last cutting could not be removed from the field. A portion of the nitrogen in the grass decomposed and became available for leaching but is not included in the Table 10 calculations.
- The amount of nitrogen mineralized from soil organic matter is difficult to quantify and can be substantial (Sullivan, 2013).
- The organic matter concentration used in the PAN estimates for past years was based on the mean organic nitrogen concentration in manure during the study and may be biased either high or low.
- Atmospheric deposition at the site could be higher or lower than the rate used, which would bias the N_{Excess} results either high or low.
- The amount of nitrate lost to leaching during the growing season following irrigation or heavy precipitation events may have been significant.

2. Estimate of annual nitrate residual using soil sampling results

Although the results show that the timing of sample collection can have a significant effect on the observed result, direct measurement of soil nitrate concentrations at the end of the growing season is a standard industry method for estimating residual nitrogen.

Equation 19 was used to convert fall soil nitrate sample concentration results from mg/kg dry weight to residual nitrate mass values in units of lb/acre. To make this conversion, Equation 19 requires a value to be set for the bulk density of the soil (in g/cm^3). The bulk density value for Hale silt loam soil (0- to 10-inch depth) at the site is 1.13 g/cm^3, assuming the average organic matter content at the site is 7.5% (average of annual values at the site). Therefore, the conversion factor from mg/kg nitrate dry weight to lb/acre ($\rho_b \ x \ 2.791$) is 3.1. This is somewhat lower than the conversion factor suggested for western Washington (3.5) by Sullivan and Cogger (2003).

$$RN_{Soil} = C_{Soil\ N}\ x\ \rho_b\ x\ 2.79 \qquad (Eq.\ 19)$$

where:
RN_{Soil} = Residual soil nitrate mass at 1-foot depth (lb/acre)
$C_{Soil\ N}$ = Soil nitrate concentration, dry weight (mg/kg DW)
ρ_b = Soil dry bulk density (g/cm^3)
2.791 = Unit conversion constant (to convert mg NO_3-N/kg (dry weight) to lbs NO_3-N/acre-ft)

Table 11 presents both the average and maximum soil nitrate concentration measured at the site each year from September 1 through October 31 (2004-2008). The corresponding residual mass values estimated using Equation 19 are also presented. These residual mass values represent the amount of nitrate that end-of-season soil sampling in the top 1 foot indicated was potentially available to leach to groundwater. The residual N_{Excess} mass estimates determined by the Method 1 mass balance evaluation are also presented in Table 11 for comparison.

Table 11. Average and maximum end-of-season soil nitrate concentrations and estimated residual mass values.

Study Year	Sept-Oct average soil nitrate concentration ($C_{Soil\ N\ AVG}$) (mg/kg)	Nitrate mass residual predicted by average fall soil concentration ($RN_{Soil\ AVG}$) (lbs/acre)[1]	Sept-Oct maximum soil nitrate concentration ($C_{Soil\ N\ MAX}$) (mg/kg)	Nitrate mass residual predicted by maximum fall soil concentration ($RN_{Soil\ MAX}$) (lbs/acre)[1]	Method 1 mass balance residual nitrogen mass estimate (N_{Excess}) (lbs/acre)[2]
2004	28.0	87	43.0	133	N/A
2005	19.9	62	30.0	93	87
2006	20.9	65	29.0	90	-79
2007	16.4	51	25.3	78	-78
2008	30.7	95	41.2	128	146
Average	23	72	34	104	19

[1] Estimates calculated using Equation 19. Soil dry bulk density assumed = 1.13 g/cm^3.
[2] Values from Table 10.

The 4-year average nitrate residual mass values of 72 and 104 lbs/acre estimated from soil samples are approximately 4 to 5 times greater than the 4-year average estimated by the mass balance approach (19 lbs/acre). It is important to note that the root mass of a growing crop also has demand for nitrogen, although we do not have good estimates for this number. This root mass has a growth and death cycle which leads to release of nitrogen in soil.

The nitrate residual mass values in Table 11 are predicted using soil sample results only from the September 1 to October 31 time frame. In some cases, soil nitrate concentrations measured after October suggest significantly higher nitrate residuals. For example, in 2006, the predicted September to October $RN_{Soil\ MAX}$ value was ~90 lbs/acre. However, if the maximum soil measurement collected in November of that year (60 mg/kg) had been used for the calculation, the resulting $RN_{Soil\ MAX}$ estimate would increase to ~186 lbs/acre. This value is more than twice that calculated with the September to October data and would be nearly 3 times as great as the $RN_{Soil\ AVG}$ value.

The annual nitrate residual mass values predicted by soil sampling are not consistent with the values predicted by the mass balance approach. In some cases, predictions were very close (the 2005 maximum soil nitrate residual vs. the 2005 mass balance predicted residual) as shown in Table 11. In other cases, the estimates were very different (e.g., 2006 and 2007).

Correlation of annual nitrogen residual estimates with groundwater quality conditions

Mass balance nitrogen residual estimates vs. groundwater nitrate concentrations

In order to observe how well the mass balance analysis approach predicted the near-term effect of end-of-season nitrate leaching to groundwater, we compared the annual N_{Excess} to the average late fall/early winter (November 1-December 31) nitrate concentration in the shallow monitoring wells. Despite intensive sampling of nitrogen inputs and outputs more extensive than normally conducted by dairy producers, the end-of-season mass balance nitrogen residual estimates did not correlate with groundwater nitrate concentrations during the higher recharge months immediately following the growing season ($r^2 = 0.19$; Table 12 columns B, C and D; Figure 57)

Table 12. Correlation of nitrogen mass residual estimates to mean November to December groundwater nitrate concentrations.

A	B	C	D	E	F	G	H
Year	**Nov-Dec GW nitrate-N conc. mean (mg/L)[1]**	**Residual nitrogen mass predicted by Method 1 mass balance (N_{Excess}) (lbs/acre)**	**Mass balance N_{Excess} vs. Nov-Dec GW nitrate-N conc. (r^2)**	**Residual nitrogen mass predicted by average soil nitrate conc. ($RN_{Soil\ AVG}$) (lbs/acre)**	**$RN_{Soil\ AVG}$ vs. Nov-Dec GW nitrate-N conc. mean (r^2)**	**Residual nitrate mass predicted by maximum soil nitrate conc. ($RN_{Soil\ MAX}$) (lbs/acre)**	**$RN_{Soil\ MAX}$ vs. Nov-Dec GW nitrate-N conc. mean (r^2)**
2004	26.5	NA	0.19	87	0.12	133	0.30
2005	18.2	87		62		93	
2006	10.0	-79		65		90	
2007	7.3	-78		51		78	
2008	9.2	146		95		128	

[1]Values represent the mean concentration of the 6 shallow wells (AKG721, AKG722, AKG723, AKG724, AKG725, and AKG727) between November 1 and December 31.
GW = groundwater. conc. = concentration.

The difficulty in accurately estimating all components of the mass balance (Equation 12) makes this method unreliable for predicting the effects on groundwater. The uncertainty in the "internal loading" contribution of nitrogen that occurs due to mineralization of soil organic matter is likely a primary explanation for the lack of correlation (Sullivan, 2013; Zebarth, 2013). Storage of nitrogen in the root system of the crop could be another influential factor.

A mean value for multiple annual mass balance residuals (i.e., 3 years) may produce a result closer to the actual amount of nitrate leached. Although we took into account nitrate mineralization from previous years in the mass balance calculation, actual nitrate mineralization from previous manure applications may vary from the method predictions. A longer period of record with more stable inputs and outputs would be needed to test whether a 3-year average mass balance residual coincides better with shallow groundwater nitrate concentrations.

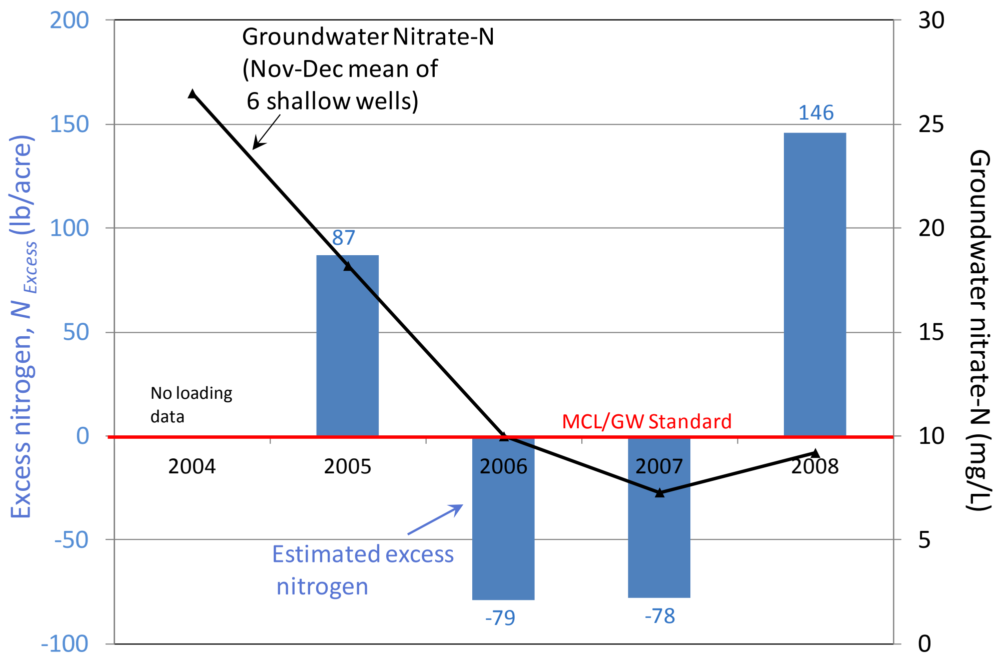

Figure 57. Comparison of annual N_{Excess} (bars) with mean winter (November-December) groundwater nitrate-N concentrations in 6 shallow wells (line).

MCL = maximum contaminant level.

Fall soil nitrate residual estimates vs. groundwater nitrate concentrations

Data were not available for nitrogen application in 2004; however, high soil nitrate concentrations in the fall (up to 160 lb/acre in August) indicated a substantial residual of nitrate was available for leaching during the fall and winter of 2004-2005 (Figure 26, Plate 2). Accordingly, the highest groundwater nitrate concentration observed, 45 mg/L-N, occurred on December 28, 2004.

While soil nitrate concentrations indicated substantial nitrate was available for leaching in the fall each year, neither the $RN_{Soil\ AVG}$ nor the $RN_{Soil\ MAX}$ values were significantly correlated with the mean November-December groundwater nitrate concentrations (Table 12, columns B, E, F, G, and H; Figure 58). This is likely because mineralization and leaching are continual processes. The concentration of nitrate in the soil can only indicate the amount left over at that point in time, with no indication of the amount of nitrate that has already leached or the amount that will become available. This suggests that fall soil nitrate monitoring, even when conducted at a high frequency, is not a reliable predictor of groundwater responses to nutrient management activities.

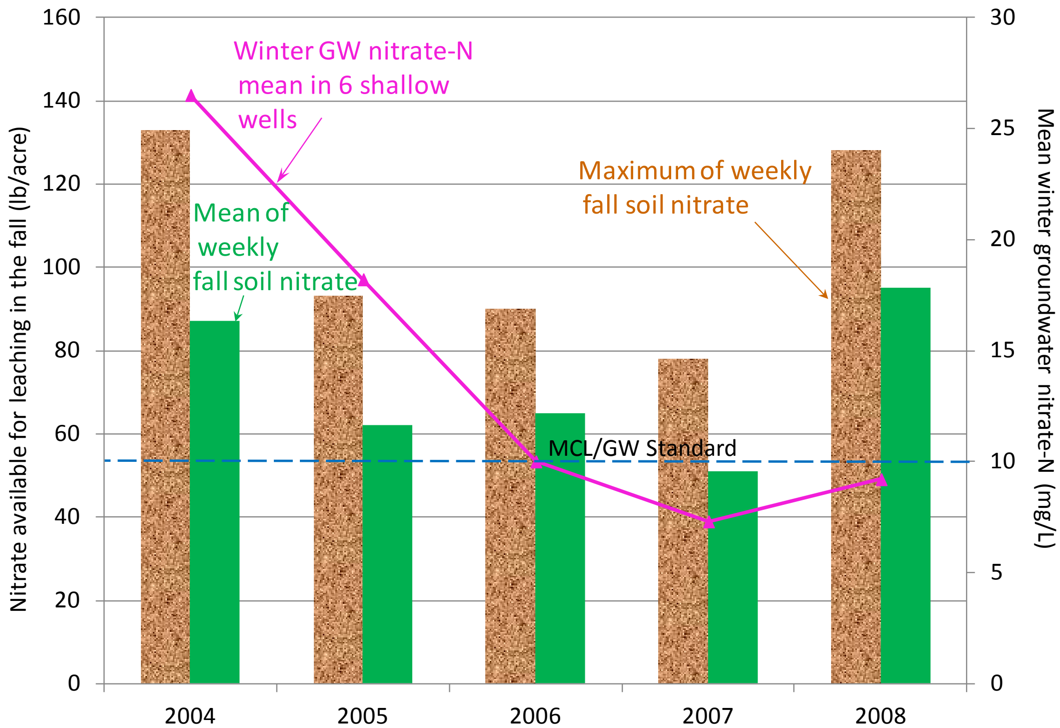

Figure 58. Comparison of annual fall soil nitrate mean concentrations (green, non-textured bars) and maximum concentrations (brown, textured bars) with mean winter (November-December) groundwater nitrate-N concentrations in 6 shallow wells (line).

Groundwater nitrate GWNO3-BACKCAST modeling

Because neither the mass balance nor the fall soil nitrate field methods provided a reliable prediction of groundwater responses to manure management, we used a simplified mixing-box model (GWNO3-BACKCAST; Pitz, 2014, in preparation) as an alternative method for estimating the annual residual nitrogen at the study field.

The BACKCAST model back-calculates the mass load of soluble nitrate (model variable name: $NO3_{TotExcess}$) required to produce a known groundwater nitrate condition (model variable name: $C_{GW\ Outflow\ NO3}$). The $NO3_{TotExcess}$ values predicted by the model are directly comparable to the Method 1 mass balance N_{Excess} residual values and also to the $RN_{Soil\ AVG}$ and $RN_{Soil\ MAX}$ residual values estimated by soil sampling.

Although the BACKCAST model is based on a simplified conceptual model of nitrate loading to an aquifer, one important advantage of the model is that it can account for *saturated zone* processes such as mixing and denitrification. These processes are not accounted for by the mass balance or soil nitrate sampling approaches. The model can be a useful tool for approximating components of the nitrogen cycle that are otherwise difficult to measure, such as mineralization.

Appendix T presents a summary of the model equations, assumptions, and input data used for each study year. Outputs and sensitivity analysis for each input parameter are also described in Appendix T.

GWNO3-BACKCAST estimates for late fall/early winter nitrate mass loading (Period A)

The BACKCAST model was used to quantitatively predict the late fall/early winter nitrate mass load necessary to generate the average shallow groundwater nitrate-N concentrations observed between November 1 and December 31 each study year (compilation of shallow monitoring well data only). A saturated zone denitrification rate of 15% was assumed for the modeling analysis on the basis of field observations (see *Nitrate: chloride ratio as an indicator of denitrification*). Table 13 summarizes the annual $NO3_{TotExcess}$ predictions for this scenario. Figure 59 graphically compares the annual Period A $NO3_{TotExcess}$ predictions to the annual residual mass estimates produced by the mass balance and maximum fall soil nitrate methods.

Table 13. Comparison of GWNO3-BACKCAST nitrate load predictions to mass balance and soil nitrate sampling residual estimates.

Year	BACKCAST Model $NO3_{TotExcess}$ Mass Load Prediction (lbs/acre)[1]		
	Period A.15[2]	Period B.15[2]	Period A.15+B.15[2]
	Late Fall/ Early Winter[3]	Late Winter/ Early Spring[4]	Entire Wet Weather Season[5]
2004	146	84	230
2005	73	49	122
2006	44	50	94
2007	23	19	42
2008	42	42	84
Average	66	49	115

[1] Assumes 15% denitrification in the saturated zone. Value represents the nitrate mass load required to generate the observed groundwater nitrate-N concentrations, under the given model assumptions.
[2] See Appendix T for details.
[3] Values based on shallow groundwater nitrate-N mean observed during November 1 to December 31.
[4] Values based on shallow groundwater nitrate-N mean observed during January 1 to March 31.
[5] Values are the sum of columns A and B (leaching that occurred September 1 to March 31).

Estimates for $NO3_{TotExcess}$ may be biased low, because we assumed that there was no nitrate loss via denitrification in the unsaturated zone. In addition, the recharge term used in the calculation may be overestimated, because runoff of precipitation was assumed to be negligible.

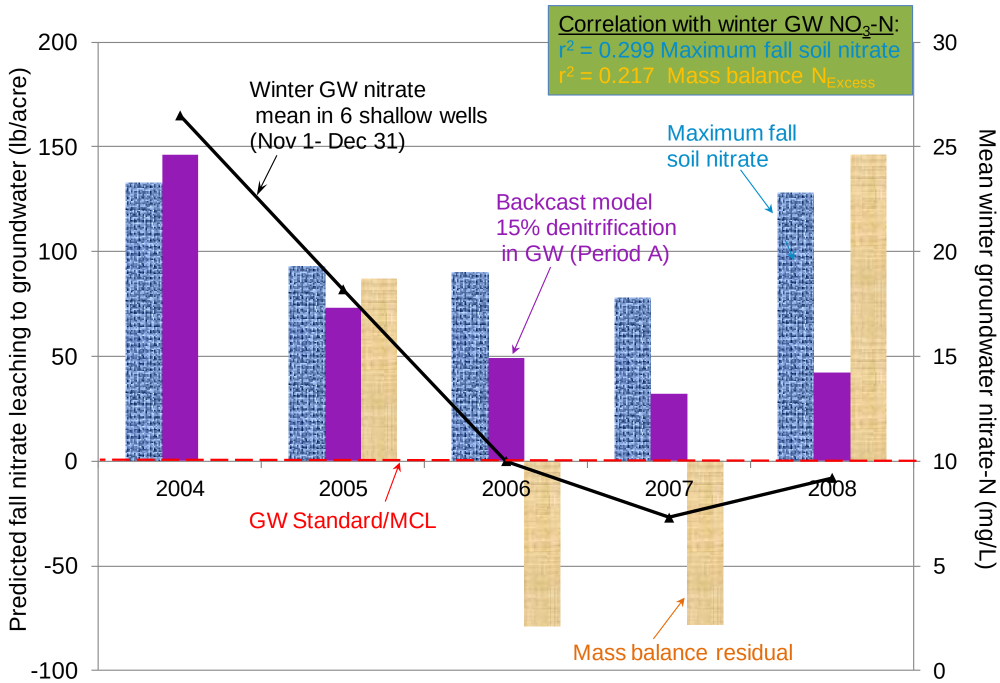

Figure 59. Comparison of Period A GWNO3-BACKCAST residual nitrate mass load predictions (purple bars) to residual mass estimates by mass balance (tan, light-textured bars) and soil nitrate results (blue, heavy-textured bars).

Key points revealed by the Period A model results include:

- $NO3_{TotExcess}$ calculated by the model for the late fall/early winter period was highest in 2004 and decreased through 2007, followed by a rebound in loading in 2008. This pattern is similar to the trend observed in the shallow groundwater nitrate concentrations (see Figure 59).
- The BACKCAST model calculated twice as much nitrate reaching the water table in the late fall/early winter of 2004 (146 lb/acre) as in the late fall/early winter of 2005 (73 lb/acre).
- The Method 1 mass balance approach estimated significant nitrogen deficits during both 2006 and 2007. The BACKCAST model, however, predicted excess nitrate mass loads occurring during the same period. A mass balance mean for multiple years (i.e., the current year and 2 previous years) may produce a result closer to the actual amount of nitrate leached and closer to the BACKCAST model results. Soil sample-derived residuals for 2006 and 2007 also indicated substantial excess nitrogen present in the soil column at the end of the growing season.
- In 4 out of 5 years, the residual nitrate mass estimated by maximum fall soil nitrate concentration was greater than the mass required by the BACKCAST model to generate the observed Period A groundwater nitrate condition. It is possible that the BACKCAST model assumptions were too conservative or that the maximum fall soil nitrate value does not represent the amount of nitrate reaching the water table due to factors such as denitrification in the soil or crop uptake after soil nitrate measurements.

- In 2008, both the mass balance and soil nitrate residual estimates indicate significantly more residual nitrogen mass was present in the soil column at the end of the growing season than the BACKCAST model $NO3_{TotExcess}$ predictions. Reasons for this discrepancy could be that the mass balance mineralization factor used was not accurate, the BACKCAST model assumptions were too conservative, or that denitrification in the soil removed a significant portion of the residual nitrate before it reached the water table.

GWNO3-BACKCAST predictions for late winter/early spring nitrate loading (Period B)

The BACKCAST model was also used to evaluate nitrate loading to the aquifer during the late winter/early spring period. This analysis was conducted to assess potential nitrate loading from ongoing mineralization beyond the fall months. For this time frame (Period B), the model was used to estimate the nitrate mass load required to generate the average shallow groundwater nitrate concentrations observed from January 1 to March 31 (compilation of shallow monitoring well data only). Similar to the Period A model analysis, a 15% saturated zone denitrification rate was assumed for the Period B evaluation.

The predicted Period B $NO3_{TotExcess}$ values are presented in Table 13. The nitrate mass loading predicted during Period B ranged from 19 to 84 lbs/acre, with an overall average of 49 lbs/acre.

Predicted nitrate loading for the entire wet-weather period (Period A+B)

The nitrate loading predicted by the BACKCAST model during Period B is *in addition to* the loading predicted to occur during Period A. To evaluate the total predicted wet-season nitrate load occurring at the site, the Period A and Period B $NO3_{TotExcess}$ values were summed for each year (Table 13; Figure 60). The nitrate mass load predictions for the entire high-recharge season (September 1 to March 31) ranged from 42 to 230 lbs/acre. The annual average $NO3_{TotExcess}$ predicted for the combined time frames was 115 lb/acre.

On average, the BACKCAST model predicted that approximately 43% of the total wet-season nitrate loading occurred between January 1 and March 31. In 2 years (2006 and 2008), the loading predicted to occur during the late winter and early spring was about the same as the loading predicted to occur during the late fall/early winter.

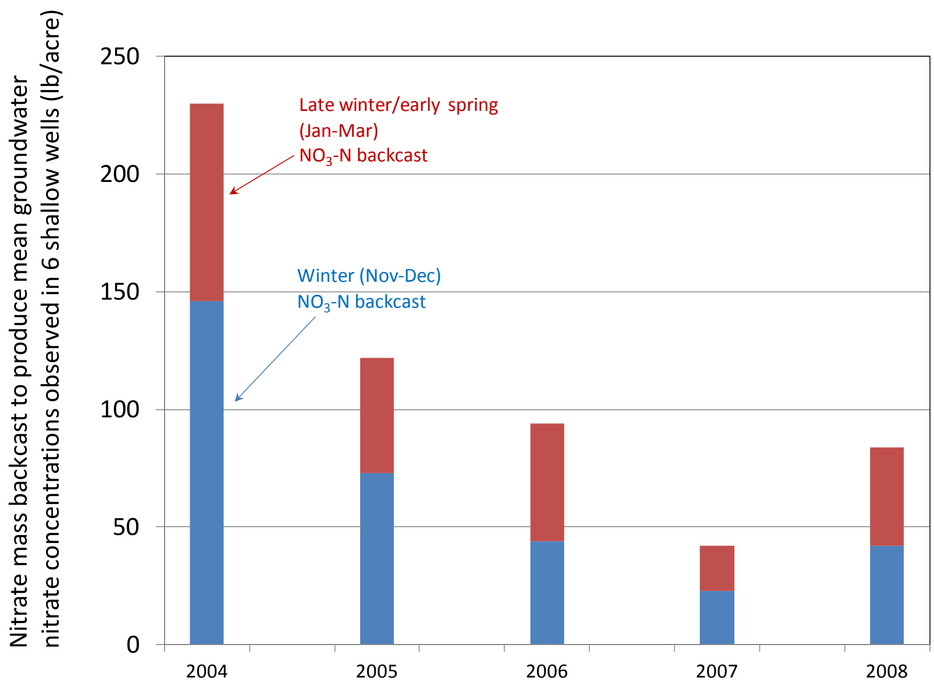

Figure 60. Total wet-season nitrate mass load predicted using the GWNO3-BACKCAST model.

Modeling implications

The BACKCAST loading estimates are approximations based on a variety of simplifying assumptions about nitrate transport and fate in the saturated zone. However, in most cases input values for the model variables were drawn directly from study data (e.g., background groundwater nitrate concentration, recharge rate, saturated zone hydraulic conductivity and hydraulic gradient, denitrification rate). The use of site-specific values, along with conservative modeling assumptions (for example, the saturated mixing zone was assumed to be only 2 feet thick), indicates that the modeled nitrate loading predictions are likely lower-bound values.

The mass balance and soil nitrate sampling approaches used in the study were intensive and more sophisticated than those typically used by dairy producers in Washington, yet these approaches were unreliable tools for predicting underlying groundwater nitrate conditions. The mass balance approach is limited by its reliance on variables that can be difficult to accurately quantify (especially internal loading contributions from mineralization). On the other hand, the study results show that the traditional post-harvest soil nitrate sampling approach can produce extreme variability in estimates of residual nitrate, depending on the time of collection. Soil sampling may also miss significant nitrate loss that occurs before samples are collected. Neither technique is well suited to account for mineralization that occurs throughout the latter half of the high-recharge season (January-March), which our analysis indicates can result in a substantial additional nitrate load to groundwater.

The poor correlation of the mass balance and soil sampling residual estimates with underlying groundwater conditions, and the BACKCAST modeling results that frequently suggest mass loading well in excess of these estimates, indicates that these techniques alone are not effective tools for managing nutrients in a manner that is reliably protective of groundwater conditions. Direct monitoring of water quality at the water table remains the most accurate and reliable method for tracking impacts of manure management on groundwater.

Soil nitrate as an indicator of leaching to groundwater

What does the recommended fall soil nitrate guideline value mean for groundwater nitrate?

The post-harvest soil nitrate test (PSNT) target value, 15 mg/kg (Sullivan and Cogger, 2003), can be translated into a potential, although not highly reliable, amount of leachable nitrate when combined with seasonal recharge. This calculation is based on the fact that 27 lb of nitrogen, when mixed with 1 acre-foot of water, is equivalent to the groundwater MCL and Washington State groundwater quality standard of 10 mg/L-N of nitrate[10] (Harter, 2012). As mentioned previously, nitrate measured in the soil at 1-foot depth indicates only the amount remaining at that point in time and is typically considered an underestimate of the total amount actually leaching to groundwater (Harter, 2013; Zebarth, 2013; Osnoy et al., 2005). The amount already leached below 1 foot cannot be accurately estimated. Therefore, the fall soil nitrate values measured during the study (1-foot depth) cannot alone account for the entire residual nitrate in the soil at the end of the growing season that is likely to leach to groundwater.

Equation 20 shows a method for calculating the average hypothetical concentration of nitrate available for leaching to groundwater based on the amount of nitrogen available at the end of the growing season and the amount of fall recharge.

$$L_{NO3\text{-}N} = \frac{(RN_{Soil})}{(2.719)R_{Fall}} \qquad (Eq.\ 20)$$

where:
$L_{NO3\text{-}N}$ = Estimated leachate nitrate concentration (mg/L-N)
RN_{Soil} = Excess nitrogen as fall soil nitrate (lb/acre)
R_{Fall} = Fall/early winter recharge (September through December) (feet)
2.719 = Units conversion factor

This method assumes that all of the RN_{Soil} mixes with all of the R_{Fall} and is transported to the water table at one time with no additional nitrate subsequently added. This is, at best, a lower-bound assumption, because RN_{Soil} was highly variable over a weekly period even when duplicate samples were analyzed during the study, and there is no way to estimate the amount of nitrate that leached below the sampling zone between the soil samples (Viers et al., 2012; Zebarth, 2013).

Although Equation 20 provides a conservative estimate of the leachate concentration during the fall/early winter season, leaching can occur anytime, especially when precipitation exceeds evapotranspiration. Mineralization continues during and after the fall/early winter period, generating additional nitrate available for leaching in addition to the end-of-growing-season RN_{Soil} value (Zebarth, 2013; Zhao et al., 2010). Nitrate mineralized during the late winter/early spring

[10] Chapter 173-200 WAC

has been found to quickly leach to the water table in the nearby Abbotsford, B.C. area (Zebarth, 2013).

Table 14 shows the results of combining the estimated R_{Fall} values (Appendix J, Table J.5) with the target PSNT guideline for grass of 15 mg/kg (Sullivan and Cogger, 2003; 47 lbs/acre for the study site) using Equation 20 and assuming no additional nitrate in the second foot of soil. The range of calculated leachate nitrate concentrations for the site is 11 to 18 mg/L-N, which exceeds the Washington State groundwater quality standard for nitrate[11].

Table 14. Estimated nitrate-N concentration in fall/early winter leachate assuming that 15 mg/kg of soil nitrate in the top 1 foot at the study site is mixed with the fall/early winter recharge during the study.

	2004	2005	2006	2007	2008	Average
Fall recharge, R_{Fall} (feet)[1]	1.57	1.19	1.39	0.98	1.32	1.29
Nitrate-N concentration in water with 15 mg/kg soil nitrate mixed with R_{Fall} *(bulk density conversion= 3.1)*	11	14	12	18	13	14

[1] September 1 through December 31.

The amount of residual fall soil nitrate that, when mixed with the average observed fall recharge, 1.3 feet, would be equivalent to 10 mg/L is 35 lb/acre (11 mg/kg). This is a small amount of nitrate relative to the accuracy of the PSNT.

Fall soil nitrate variability and sample timing

Results for PSNT sampling can be highly variable during the period recommended for sampling (after the last harvest and before 5 inches of fall precipitation; Figure 26, Plate 2). Factors such as timing of manure application, temperature, the amount and timing of recent precipitation, and inherent heterogeneity of nitrate in soil all affect the concentration of nitrate in a soil sample in the fall on a given day (Sullivan and Cogger, 2003; Oenema et al., 2010).

The range for weekly fall soil nitrate concentrations from September 1 to October 31 each year was 12 to 43 mg/kg (37 to 133 lb/acre; Table 15). The large variation in soil nitrate concentrations during the fall in the 0 to 1-foot depth interval illustrates that a single sample is unlikely to correspond with the actual total amount of nitrate leaching to groundwater.

Table 15 also shows the results of combining R_{Fall} values from Table 14 with either the minimum or maximum fall soil nitrate concentrations using Equation 20. The annual range in variation between the highest and lowest soil nitrate-derived leachate concentration was 12 to 18 mg/L-N.

[11] Chapter 173-200 WAC

Table 15. Fall soil nitrate variability and resulting soil nitrate-derived leachate concentrations for soil samples collected from September 1 through October 31.

Year	Minimum soil nitrate (mg/kg)	Maximum soil nitrate (mg/kg)	Difference between minimum and maximum soil nitrate (mg/kg)	Fall recharge (feet)	Minimum estimated leachate nitrate concentration (mg/L-N)[1]	Maximum estimated leachate nitrate concentration (mg/L-N)[1]	Difference between minimum and maximum leachate estimates (mg/L-N)
2004	19	43	24	1.57	14	31	18
2005	12	30	19	1.19	11	29	18
2006	16	29	14	1.34	13	25	12
2007	12	25	14	0.98	14	30	16
2008	21	41	20	1.32	18	36	18

[1] This assumes that recharge mixes completely with the soil nitrate and that 27 lb/acre of nitrogen in 1 acre-foot is equivalent to 10 mg/L nitrate-N.

Reasons for high seasonal variability in fall soil nitrate measurements include combinations of the following:

- Inherently high spatial variability of nitrate in soil and over short time periods.
- Timing of the last harvest (which requires a period of dry weather).
- Re-wetting of soil in the fall after drying all summer, which causes a surge in mineralization; percolating water carries recently mineralized nitrate to groundwater.
- Timing of the last manure application (sometimes after the last harvest).
- Crop uptake and removal, which depends on complex interactions between temperature, soil moisture, precipitation, and irrigation.

In most years, when the maximum fall soil nitrate concentration substantially exceeded 15 mg/kg (2004, 2006, and 2008), winter groundwater nitrate concentrations increased in most, if not all, monitoring wells (Figure 40, Plate 5). In contrast, during the winter following the lowest maximum fall soil nitrate concentration (2007), nitrate decreased in some shallow wells, and increases in other wells were lower than in other years.

Estimated leachate nitrate concentrations based on mean and maximum fall soil nitrate concentration

If the average 1-foot fall soil nitrate concentration for each year were mixed with the recharge for the season (Equation 20), the predicted leachate nitrate concentration would be 16 to 23 mg/L-N as shown in Table 16. If the maximum fall soil nitrate concentration were combined with wet-season recharge, the annual predicted leachate nitrate concentration would be 29 to 50 mg/L-N. (These calculated concentrations include soil nitrate values from November 1-15, 2 weeks more than the previous fall soil nitrate analyses, to account for any nitrate that might have affected groundwater after the normal soil sampling period.)

Table 16. Estimated leachate concentration if the fall soil nitrate at 1-foot depth during the study (mean or maximum from September 1 - November 15) were mixed with the fall/early winter (September-December) recharge observed.

	2004	2005	2006	2007	2008
Fall/early winter recharge (feet)[1]	1.57	1.19	1.39	0.98	1.32
Mean fall soil nitrate (lb/acre)[2]	82	52	77	48	81
Estimated leachate nitrate (mg/L-N)[3]-mean	**19**	**16**	**21**	**18**	**23**
Maximum fall soil nitrate (lb/acre)[2]	133	93	186	78	128
Estimated leachate nitrate-N (mg/L)-max	**31**	**29**	**50**	**29**	**36**

[1] September 1 through December 31
[2] September 1 through November 15
[3] Based on Equation 20

Conclusions

Intensive monitoring of soil, manure, crop, and groundwater showed that management practices at a manured dairy grass field over the Sumas-Blaine Aquifer (SBA) resulted in mean shallow groundwater nitrate concentrations of 5.5 to 30 mg/L-N. Fifty-six percent of monthly mean groundwater nitrate results were above 10 mg/L-N. Mean groundwater nitrate results were below 10 mg/L-N following application of manure nitrogen at rates close to crop removal rates and at appropriate times.

Nitrate concentrations in shallow groundwater underlying the study field generally declined over the first 3 years of the study (2005-2007) from an average concentration in 6 shallow monitoring wells of 30 mg/L-N (maximum of 45 mg/L-N). This decline is interpreted to be primarily the result of a steady reduction in the amount of nitrogen applied to the field. In the fourth year (2008), groundwater nitrate concentrations increased presumably due to an excess of manure nitrogen inputs relative to outputs. The annual mass of nitrogen removed in the crop was consistent throughout the study and did not correspond with the large variation in the annual mass of nitrogen applied (394-715 lb/acre).

Factors affecting nitrate levels in groundwater and soil

Many factors affected groundwater and soil nitrate concentrations relative to manure applications. Factors related to environmental and management practices act simultaneously and in complex ways. These factors resulted in high spatial and temporal variability in groundwater and soil conditions.

The main environmental factors that influenced conditions in the study include:

- Hydrogeologic conditions
  - The shallow groundwater elevation in winter led to rapid transport of nitrate to groundwater.
  - We found evidence of substantial loss of nitrate to nitrogen gas in 4 out of 6 monitoring wells via microbially-mediated denitrification. Denitrification is less likely to occur in coarse-grained materials.
- Precipitation
  - Most of the annual precipitation in the area occurs during the period of limited crop growth (October through March). Precipitation that infiltrated through the soil to the groundwater (recharge) carried available nitrate to the water table.
- Soil conditions

  Summertime drying, which we observed in the beginning of the study, may have slowed mineralization and dampened crop uptake. This in turn may have increased the amount of residual nitrate available to leach to groundwater in the fall.

Management practices that influenced conditions in the study include:

- Rate of nitrogen application (external loading): The more nitrogen applied to the field in excess of the crop demand, the higher the amount of nitrate that reached the water table. Higher shallow groundwater nitrate concentrations occurred in the early winter following nitrogen application in excess of crop uptake (2005, 2008). Groundwater nitrate concentrations were lower (and mostly below 10 mg/L-N) when the amount of nitrogen applied was less than crop removal (2006, 2007).
- Rate of nitrogen mineralization (internal loading): Organic nitrogen in the soil from previous years, and manure organic nitrogen from the current year, are bacterially converted to plant-available nitrate, but accurately quantifying the amount and timing is difficult. The average estimated annual amount of mineralized nitrogen available at our study site was roughly one-half of the annual amount of plant-available nitrogen. Mineralization contributed to groundwater nitrate loading throughout the high-precipitation season.
- Tillage effects: Tillage of the field during the first year (2004) led to elevated nitrogen mineralization in the soil. This contributed to groundwater nitrate loading at the beginning of the study but was not the dominant factor.
- Timing of manure applications:
  - Manure application late in the growing season (or after the growing season), when recharge was high, was followed by elevated groundwater nitrate concentrations.
  - Manure application too early in the spring was followed by elevated groundwater nitrate concentrations.
- Denitrification: Denitrification in shallow groundwater at the site was controlled by reduced dissolved oxygen concentrations in 4 out of 6 wells. We estimated that an average of 28% of the nitrate in 4 wells with at least occasionally low dissolved oxygen levels was converted to nitrogen gas. Denitrification was more prevalent in the summer and fall, when recharge was low, than in the winter.

Nitrogen residual estimates compared with groundwater nitrate concentrations

An important goal of the Dairy Nutrient Management Plans (DNMPs) is to minimize the amount of residual nitrogen left in the field after the growing season and available to leach to groundwater.

Mass balance and post-harvest soil nitrate residual estimates

We evaluated 2 field-based methods for estimating the amount of residual nitrogen left in the study field at the end of each growing season:

- Mass balance estimate of annual nitrogen residual
- Fall soil nitrate residual (1 foot depth)

The mass balance analysis used the most recent Oregon State University Extension Service method for estimating plant-available nitrogen. The annual nitrogen residual from 2005 to 2008 was -79 lb/acre to 146 lb/acre with a 4-year average of 19 lb/acre. Significant nitrogen deficits were calculated for 2006 and 2007. The annual mass balance residual did not correspond well with the nitrogen application rates. The main source of uncertainty in the mass balance evaluation was probably the contribution from mineralized organic matter from past years.

The second method used the fall soil nitrate values collected weekly in September and October to estimate the annual mean and maximum soil nitrate residual. The 4-year average fall soil nitrate result was 72 lb/acre; the maximum was 104 lb/acre. These averages were 4 to 5 times the average calculated by the mass balance method, and have high variability over short time periods. Soil nitrate residual values did not track closely with nitrogen application rates.

Fall soil nitrate results are an underestimate of the amount of nitrate available for leaching at the end of the growing season, because additional nitrate in the soil at 2 feet or deeper was not included in the calculation as well as any nitrate that leached before or after the sample was collected.

If the average 1-foot deep fall soil nitrate concentration for each year were mixed with the recharge for the fall/early winter season, the predicted leachate nitrate concentration would be 16 to 23 mg/L-N. If the maximum fall soil nitrate concentration for each year were combined with recharge, the annual predicted leachate nitrate concentration would 29 to 50 mg/L-N.

We attempted to correlate results of these 2 nitrate residual estimate methods with early winter groundwater nitrate concentrations, because early winter is typically the time when most of the end-of-season residual nitrate in the soil reaches the water table. Neither approach correlated well with groundwater nitrate concentrations that we observed in our study.

The nitrogen mass balance method contains many variables with inherent uncertainty, especially nitrate mineralized from past manure applications. This high uncertainty may explain the lack of correlation with groundwater concentrations. The high variability of soil nitrate conditions during the post-harvest soil sampling period likewise contributed to the unreliability of the soil nitrate method for predicting early winter groundwater nitrate concentrations. In addition, neither the nitrogen mass balance method nor the fall soil nitrate method takes into account processes that occur in the groundwater, such as mixing and denitrification.

GWNO3-BACKCAST model

An alternative method for estimating residual nitrate, the GWNO3-BACKCAST model, predicted the amount of nitrate needed to create the groundwater nitrate concentrations observed. We used this method, because the mass balance and soil nitrate methods were not good predictors of nitrate leaching. The model is based mainly on parameters measured during the study: recharge rate, groundwater nitrate, hydraulic conductivity and horizontal gradient.

In this study, on average the BACKCAST model predicted an average nitrate loading to groundwater during the wet season (September-March) of 115 lb/acre, with a range of 42 to 230 lb/acre. Of the total predicted nitrate loading to groundwater, 66 lb/acre occurred during the late fall/early winter and 49 lb/acre during the late winter/early spring. The BACKCAST model predictions were most sensitive to the recharge rate, the denitrification rate in groundwater, and the thickness of the groundwater mixing zone.

Post-harvest soil nitrate test

We evaluated the recommended post-harvest soil nitrate concentration for dairy grass fields in western Washington, 15 mg/kg, in relation to the recharge amounts observed. The calculated leachate nitrate concentration that would result from combining this soil nitrate concentration with the observed fall/early winter recharge volumes ranged from 11 to 18 mg/L-N. This does not include nitrate available for leaching below the top foot of soil. All soil nitrate samples collected according to the recommended protocols for post-harvest soil nitrate testing during the study exceeded 15 mg/kg.

If the seasonal *average* September to mid-November soil nitrate concentrations was used in the same calculations with the recharge that occurred during the study, the estimated nitrate concentration in leachate would have been 16 to 23 mg/L-N. If the seasonal *maximum* September to mid-November soil nitrate concentration was used in the calculation, the estimated leachate nitrate concentration would have been 29 to 50 mg/L-N.

Recommendations

Reducing nitrate leaching to groundwater at manured dairy fields over the Sumas Blaine Aquifer (SBA), like the one in this study, will require improving manure application based on evolving science and technology. This includes fine-tuning nitrogen loading analyses and taking groundwater into account. Groundwater monitoring will be needed to evaluate the effectiveness of measures to reduce nitrate loading to groundwater.

Based on the results of this study, the following actions are recommended to promote improvements to groundwater quality in the SBA and in other areas of Washington State with similar conditions.

Reduce nitrate loading to groundwater

We recommend that stakeholders develop a process whereby (1) manure and fertilizer nitrogen inputs and outputs are tracked on a field-by-field basis and (2) the information is used to minimize nitrate leaching below the root zone. Because of high seasonal rainfall and shallow depth to water, appropriate timing and amount of nutrient application is crucial. Involvement of state and local organizations, in partnership with universities, dairy and other agricultural producers, is needed to improve nitrogen use efficiency and to protect groundwater quality.

Key challenges for local conditions include:

- Availability of management practices to prevent nitrate leaching.
- Knowledge of management practices to prevent nitrate leaching.
- Part of the growing season overlaps with the groundwater recharge season.

Lessons learned in this study that could decrease nitrate leaching to groundwater in the SBA include:

- Review available mass balance assessment methods for use in Dairy Nutrient Management Plans (DNMPs). Develop or adapt existing methods so they more accurately account for effects on groundwater (i.e., more accurate assessment of soil organic matter contribution).
- Calculate nitrogen applications (manure, fertilizer, irrigation water) and crop removal based on measured amounts and nitrogen analyses. This is especially important in areas where groundwater nitrate already does not meet the drinking water standard. Timing of nitrogen application relative to recharge is especially important.
- Our results indicate that it is best to schedule the last manure application by late August to early September. Manure application during the high-recharge period (September through mid-March) is likely to increase nitrate leaching to groundwater.
- Where groundwater is well-oxygenated and denitrification rates are low, take special care to prevent overapplication or application during the high-recharge period.

- If soil moisture is low in the summer, consider irrigating to increase mineralization and nitrification and increase available nitrate for the crop. Avoid overapplication of irrigation water to prevent nitrate leaching.
- Consider extending the time between tillage events to decrease the amount of nitrogen reaching groundwater.
- Consider limiting manure application to forage crops during the first season following tillage.
- Consider updating the post-harvest soil nitrate test (PSNT) guidance to incorporate hydrologic influences (i.e., local aquifer recharge) based on the expertise of land grant universities as well as local and state scientists.
- Until revised, use the existing post-harvest soil nitrate protocols (methods and timing) and targets (15 mg/kg for grass; 20 mg/kg for corn) as criteria for evaluating manure management at dairy operations in western Washington.

Monitor to evaluate the effectiveness of management improvements

A program is needed to determine how well current and future manure management practices are working to improve groundwater quality. Because there is no reliable substitute, direct groundwater monitoring using dedicated monitoring wells is a key component of an effectiveness monitoring program.

Although groundwater monitoring is the only available way to determine the amount or the concentration of nitrate that actually reaches the water table, soil nitrate monitoring in the fall is a necessary tool for on-farm nutrient management. If conducted with limitations in mind, soil nitrate monitoring also can serve as a screening tool for closer inspection of groundwater conditions.

Nitrogen mass balance evaluations are also an important tool for dairy producers to manage nutrients and identify potential courses of action to address high soil and groundwater nitrate concentrations. Current methods for analyzing mass balances for DNMPs should be evaluated to ensure the methods are as accurate as possible.

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Appendices

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Appendix A. Glossary, Acronyms, and Abbreviations

Glossary

Aerobic: In terms of liquid water, the state that contains oxygen at a measurable level.

Ammonium: A positively charged ion that is available to plants and typically comprises a large portion of fresh manures. The symbol for ammonium is NH_4^+

Anaerobic: In terms of liquid water, devoid of oxygen at a measurable level.

Cation: A positively charged ion.

Coefficient of curvature: A number used to classify materials as sands or gravels based on the particle size distribution of a soil sample. The number is based on the particle diameters corresponding to 10, 30, and 60%, respectively passing on a standard cumulative particle size distribution curve.

Denitrification: The bacterial or chemical process whereby nitrate is converted to nitrogen gas, usually in a reducing/low dissolved oxygen environment.

Depth to water: The distance from a measuring point (i.e., the ground surface) to the saturated zone, usually measured in a well or piezometer.

Effective grain size: Also called effective diameter, d_{10}, is the particle diameter corresponding to 10% finer on the grain-size curve.

Evapotranspiration: The sum of evaporation and transpiration. Evaporation is the loss of water to the atmosphere from the ground surface down to the capillary fringe of the water table. Transpiration is the evaporation of water from plant leaves.

Groundwater: Water in the saturated zone that is under pressure that is equal to or greater than atmospheric pressure.

Hydraulic conductivity (K_H): Rate at which water moves through a material at a unit gradient and depends on the size and arrangement of the pores between the particles.

Hydrogeologic: Relating to the scientific study of the waters of the earth, especially with relation to the effects of precipitation and evaporation upon the occurrence and character of water in streams, lakes, and on or below the land surface.

Leachate: Water that percolates through a porous media carrying dissolved substances as it moves, typically to the groundwater.

Mass balance: A tool for estimating nutrient budgets that include inputs, outputs, and residuals for a system (e.g., nitrogen in an agricultural field).

Maximum contaminant level (MCL): A regulatory limit set by the U.S. Environmental Protection Agency (EPA) for contaminants in drinking water. If an MCL is exceeded, regulatory action is required under the Safe Drinking Water Act.

Methemoglobinemia: A serious health condition that reduces the ability of red blood cells to carry oxygen that can result from exposure to high levels of nitrate. In most adults and children, these red blood cells rapidly return to normal. However, in infants it can take much longer for the blood cells to return to normal. Infants who drink water with high levels of nitrate (or eat foods made with nitrate-contaminated water) may develop a serious health condition due to the lack of oxygen and call "blue-baby syndrome."

Mixing zone: The top portion of an aquifer into which recharging water is mixed over a specified period of time.

Monitoring well: A cased hole drilled and completed to specifications that enable water level and water quality sampling to be conducted that are representative of conditions in the portion of the aquifer of interest.

Nitrate: A common, highly mobile, nitrogen-based chemical compound. Ammonium from manure is typically converted to nitrate by bacteria. The symbol for nitrate is NO_3^-

Nitrogen: An element that is found in all parts of the environment and in organic/living matter.

Nitrate-N: The common reference to nitrate as nitrogen in water analyses.

Nutrient: Substance such as carbon, nitrogen, and phosphorus used by organisms to live and grow. Too many nutrients in the water can promote algal blooms and rob the water of oxygen vital to aquatic organisms. Nutrients can be transported from groundwater to surface water. Nitrate is considered a nutrient.

Recharge (noun)**:** The amount of water entering the saturated zone at the water-table surface over a period of time.

Root zone: The soil zone in which crop roots are found. The thickness varies with the type of crop. The root zone typically contains both water and air and is unsaturated.

Specific capacity: A measure of the productivity of a well estimated by measuring the pumping rate (yield) and dividing by the change in the height of water in the well (drawdown)

Study period: September 22, 2004 through March 18, 2009.

Study site (also study field): 22-acre dairy located northwest of Lynden, Washington.

Unconfined aquifer: An aquifer containing water that is not under pressure; the water level in a well is the same as the water table outside the well.

Uniformity coefficient: A number used to classify materials as sands or gravels based on the particle size distribution of a soil sample. The ratio of d_{60} to d_{10} where d_{60} is the particle diameter corresponding to 60% finer on the cumulative particle size-distribution curve, and d_{10} is the particle diameter corresponding to 10% finer on the cumulative particle size-distribution curve.

Upgradient: In hydrology, an *upgradient* location is one that exhibits a larger hydraulic head in comparison to a *downgradient* location. Water flows from areas of high hydraulic head to areas of low hydraulic head. Hydraulic head is the total pressure exerted by a water mass at any given point. Total hydraulic head is the sum of elevation head, pressure head, and velocity head.

Vadose zone: The subsurface zone that starts at the land surface and contains both air and water (is not saturated).

Water table: The level in the saturated zone at which hydraulic pressure is equal to atmospheric pressure and is represented by the water level in wells that are not pumping.

Acronyms and Abbreviations

B.C.	British Columbia, Canada
BGS	Below ground surface
DO	Dissolved oxygen
DNMP	Dairy Nutrient Management Plan
DOC	Dissolved organic carbon
Ecology	Washington State Department of Ecology
GPS	Global Positioning System
GW	Groundwater
K_H	(See Glossary above)
MCL	(See Glossary above)
MEL	Manchester Environmental Laboratory
N	Nitrogen
n	Number
NH_4^+	Ammonium
Nitrate-N	Nitrate as nitrogen
NO_3^-	Nitrate
NRCS	Natural Resources Conservation Service
PAN	Plant-available nitrogen
PSNT	Post-harvest soil nitrate test
PVC	Polyvinyl chloride
RCW	Revised Code of Washington
RPD	Relative percent difference
RSD	Relative standard deviation
SBA	Sumas-Blaine Aquifer
SOP	Standard operating procedure
TPN	Total persulfate nitrogen
U.S. EPA	U.S. Environmental Protection Agency
WAC	Washington Administrative Code
WSU	Washington State University

Units of Measurement

°C	degrees centigrade
dw	dry weight
ft	feet
in	inches
kg	kilograms, a unit of mass equal to 1,000 grams.
km	kilometer, a unit of length equal to 1,000 meters.
lb	pound
mm	millimeter
mg/kg	milligrams per kilogram (parts per million)
mg/L	milligrams per liter (parts per million)
mg/L-N	milligrams per liter as nitrogen
mL	milliliters
s.u.	standard units
ug/L	micrograms per liter (parts per billion)
um	micron
umhos/cm	micromhos per centimeter, a unit of conductivity

Plate 1. Hydrogeologic data

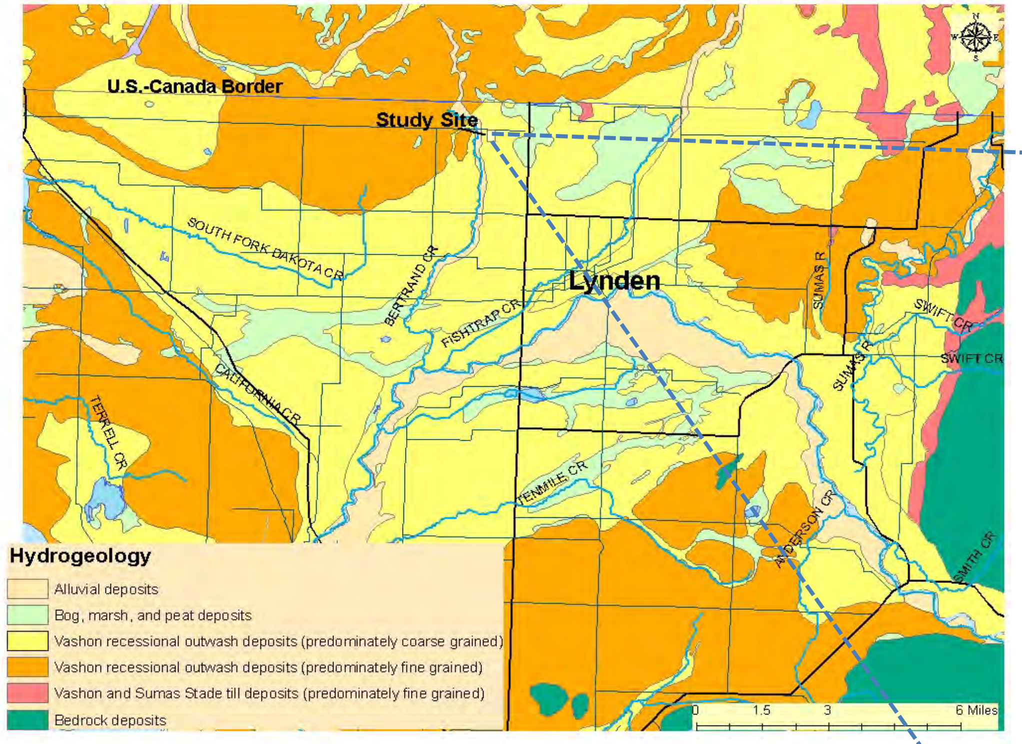
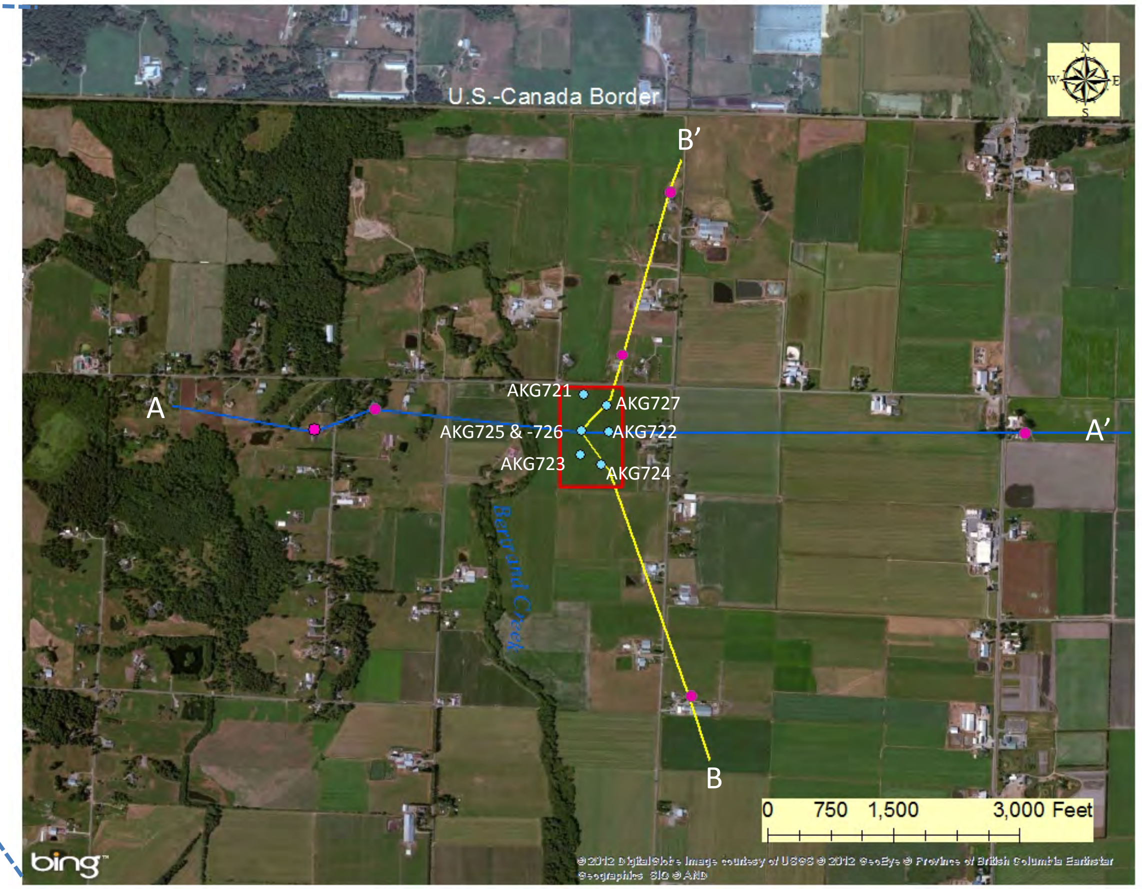


Figure 4. Local surficial geology of the study area from Jones (1999).

Figure 5. Location of wells used in Figure 6 cross-sections. The blue dots represent monitoring wells used in this study; pink dots private, domestic wells. The red line is the study area boundary.



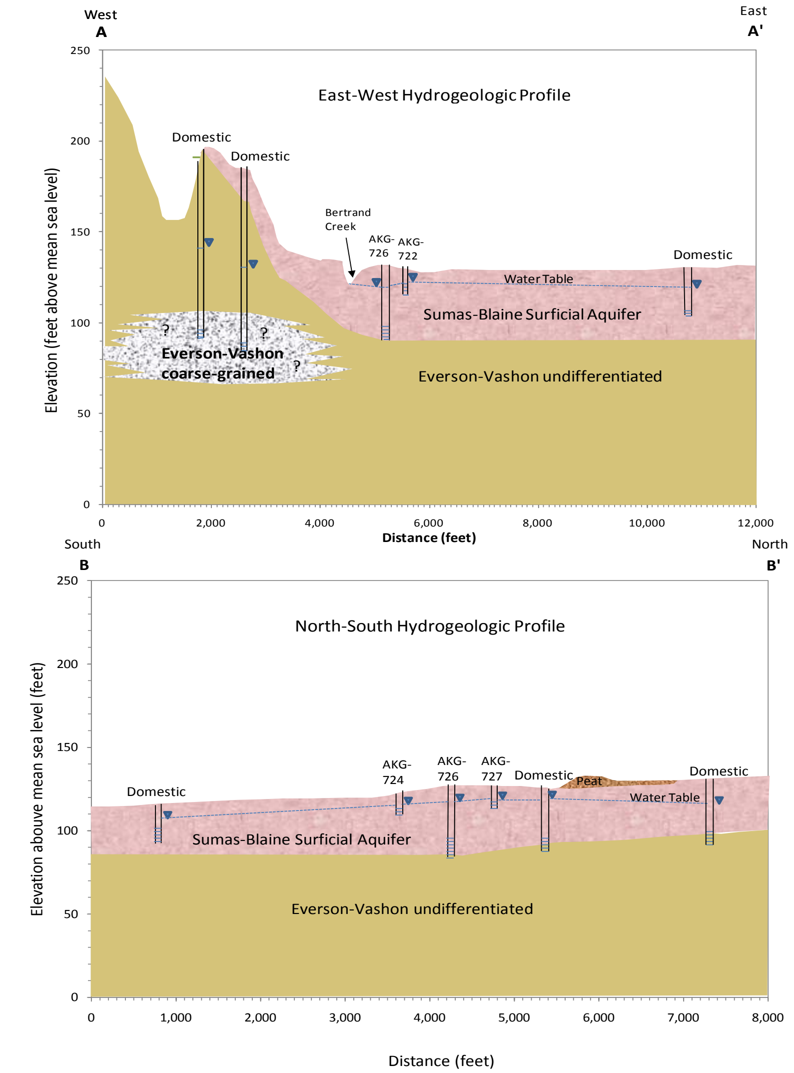


Figure 6. Generalized hydrogeologic cross-sections from Figure 5.

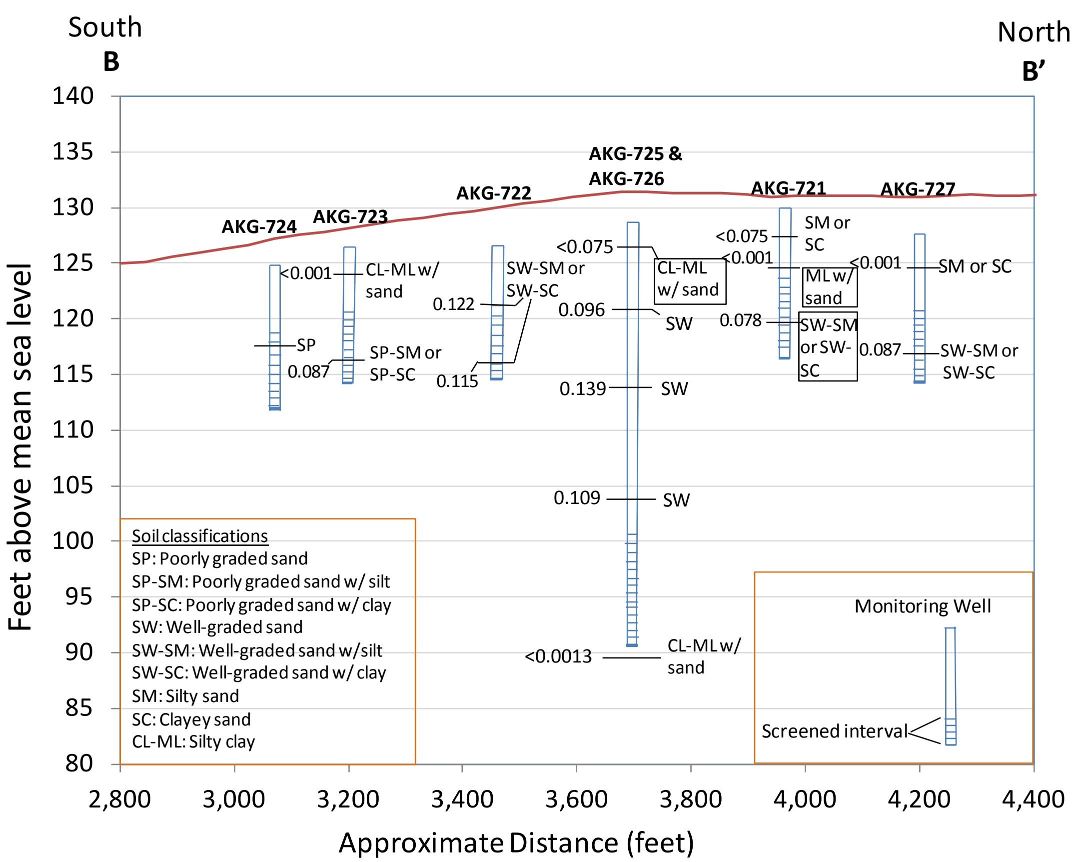


Figure 27. Soil classifications and d_{10} values for soil samples projected on land surface cross-section B-B' in Figure 5.

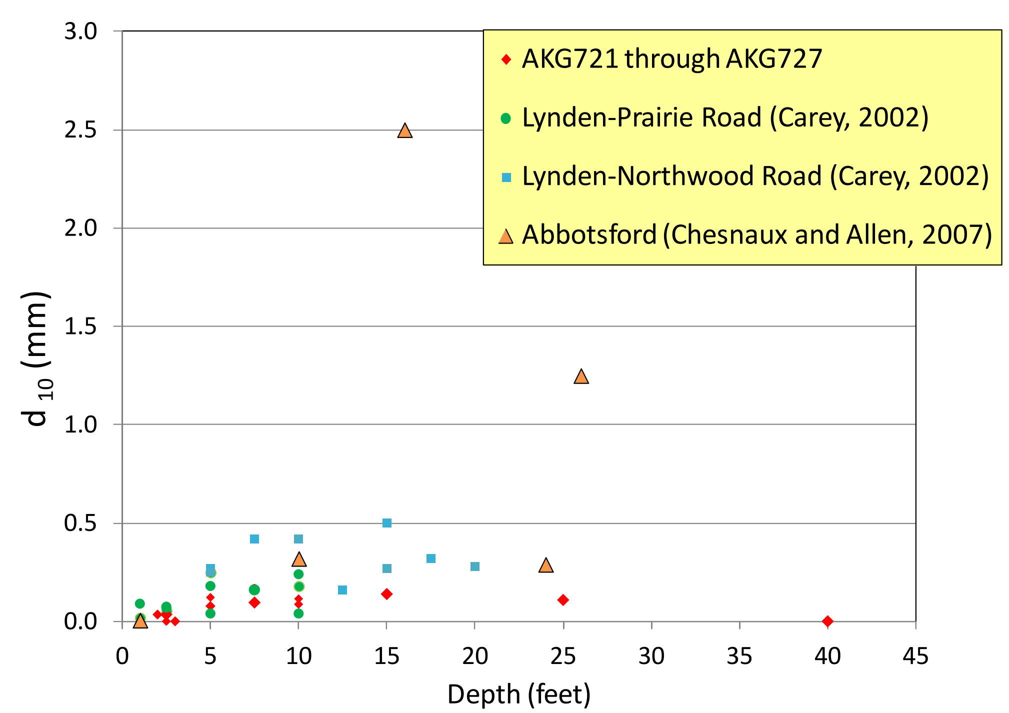


Figure 43. Effective grain size (d_{10}) values for drilling samples from monitoring wells in the U.S. and from one well near Abbotsford, B.C. See Figure 44 for locations.

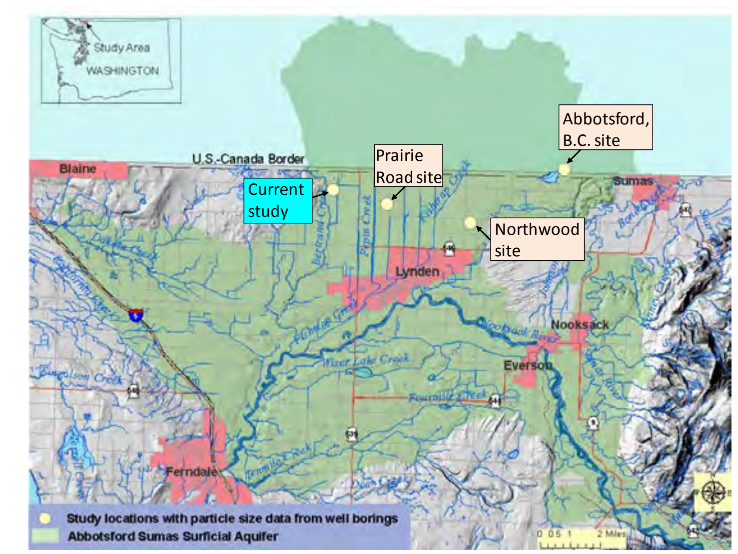


Figure 44. Locations of particle size analysis samples from well borings shown in Figure 43 (Chesnaux and Allen, 2007; Carey, 2002).

Soil Conditions

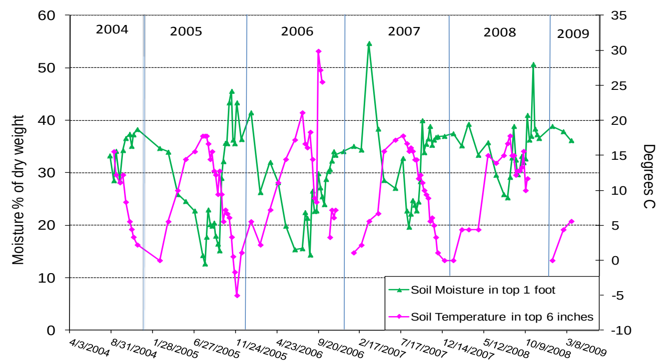


Figure 17. Soil moisture and soil temperature measurements.

Inputs of Nitrogen and Chloride

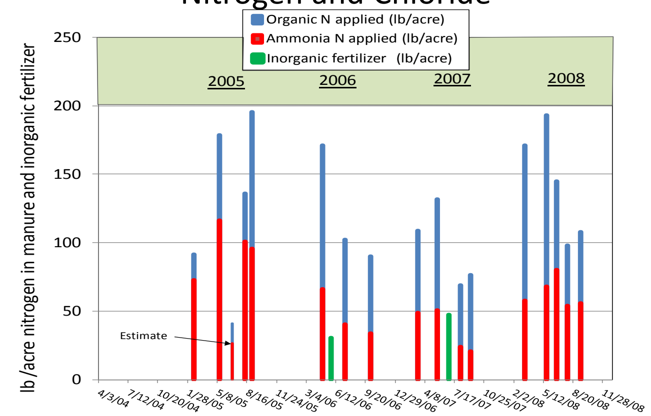


Figure 19. Total nitrogen applied in manure and inorganic fertilizer. Each bar represents one application event.

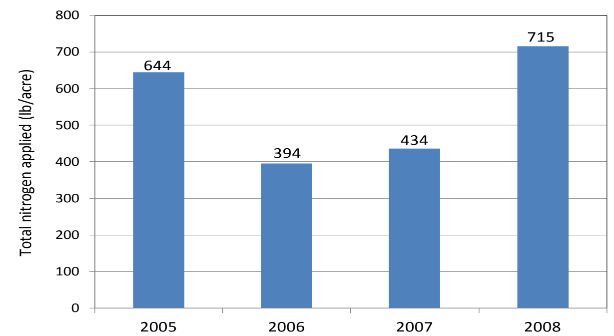


Figure 20. Total nitrogen applied annually (manure, fertilizer, and irrigation water).

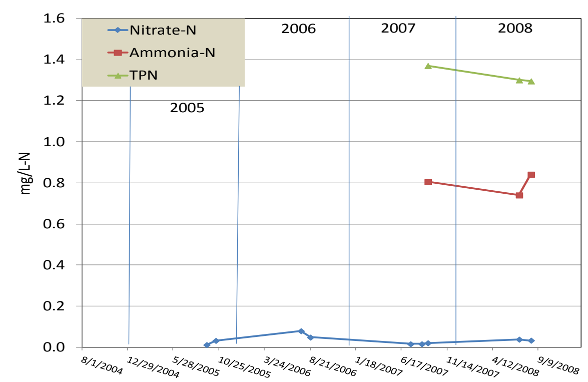


Figure 21. Concentrations of nitrate-N, ammonia-N, and total persulfate N in irrigation water applied.

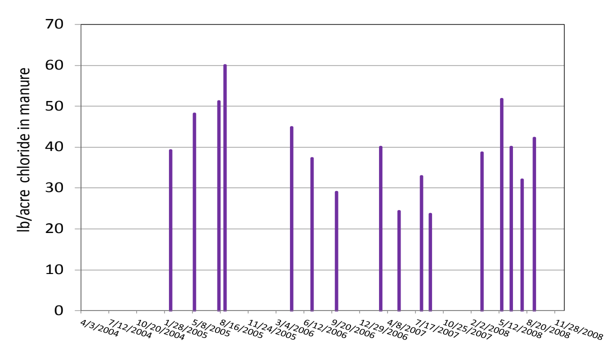


Figure 22. Chloride applied to the field in manure.

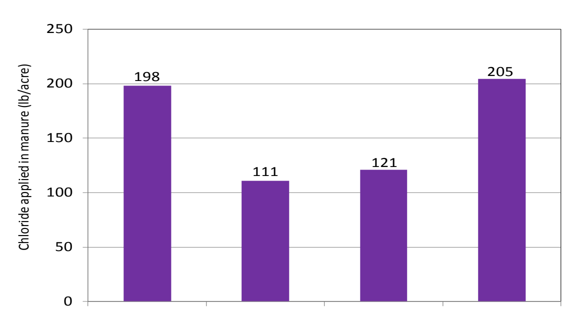


Figure 23. Chloride applied annually to the field in manure.

Outputs of Nitrogen

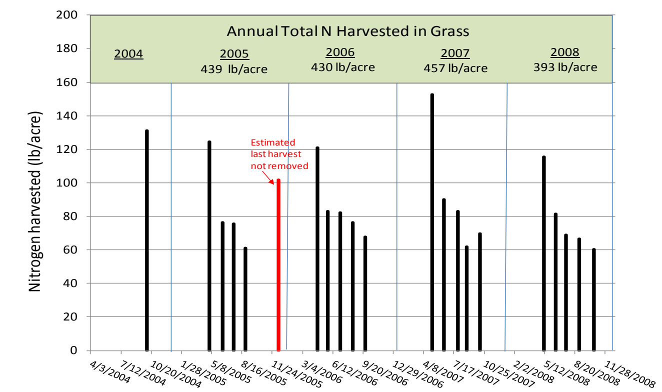


Figure 24. Total nitrogen harvested in grass for each cutting (individual bars) and annual totals for 2005 through 2008 (along the top) .

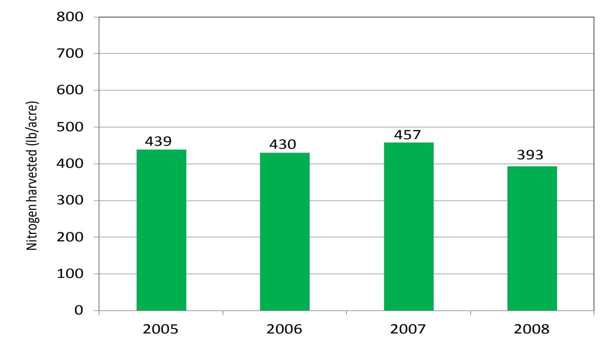


Figure 25. Total nitrogen removed in the grass harvest, including an estimate for uptake in 2005, which was not removed from the field.

Residual Nitrate

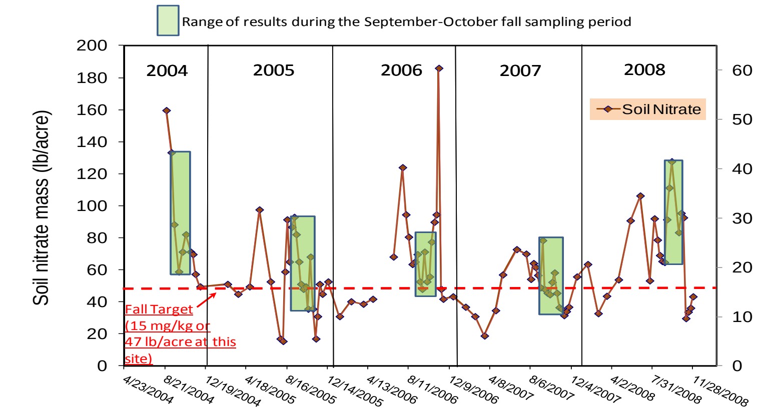


Figure 26. Soil nitrate results at 1-foot depth. Green shaded areas indicate results for the typical fall soil sampling time, September through October. Red line indicates the level below which management changes are not recommended (Sullivan and Cogger, 2003).

Plate 2. Soil conditions, nitrogen and chloride inputs, nitrogen outputs, and soil nitrogen residual.

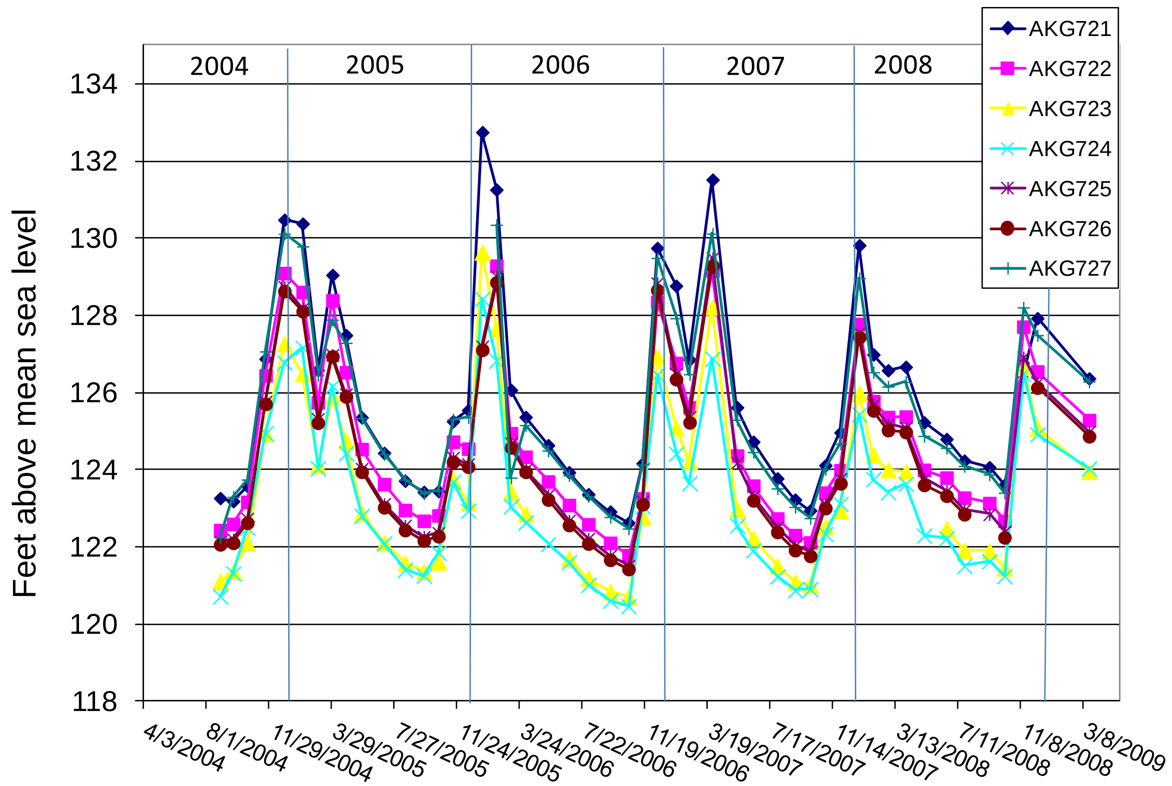


Figure 28. Water table elevations in the monitoring wells relative to the top of casing of AKG721 (134.00 feet, NAVD88) and depth to water below ground surface.

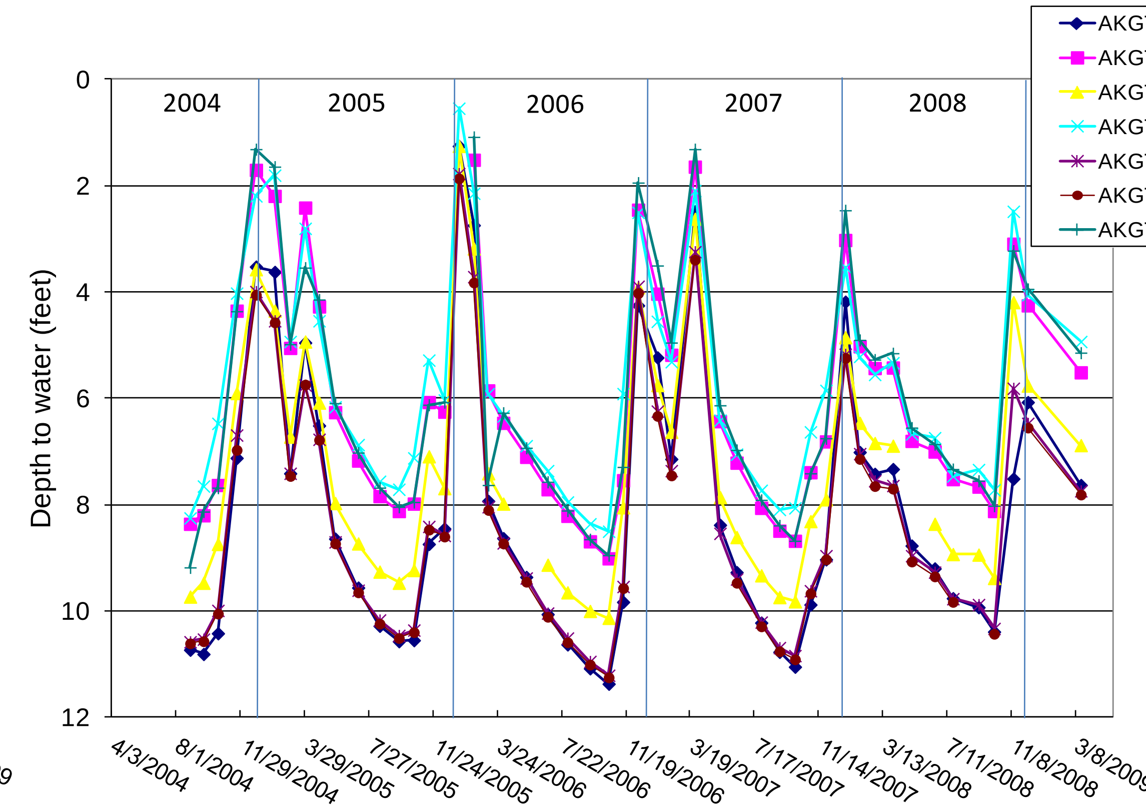


Figure 29. Depth to water below ground surface in the monitoring wells.

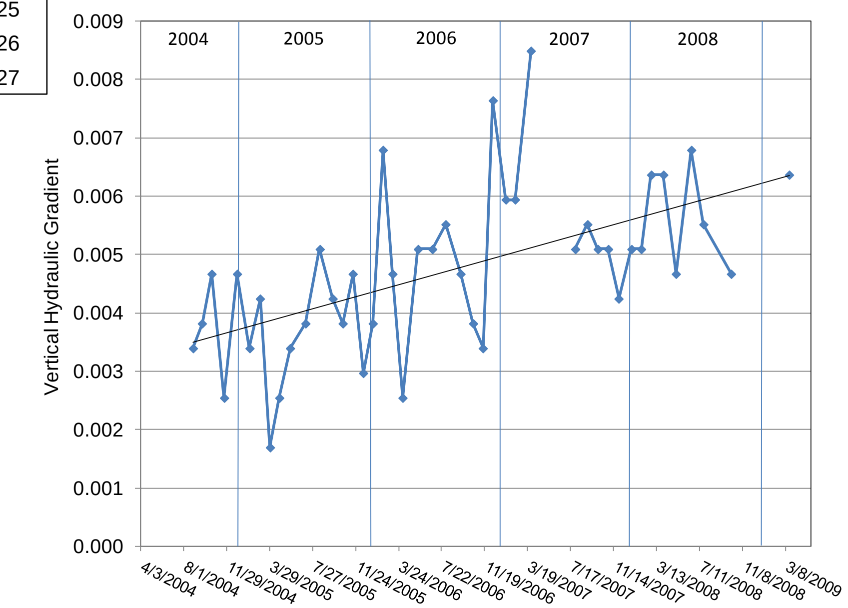


Figure 31. Downward vertical hydraulic gradient at side-by-side wells, AKG725 (13 feet deep) and AKG726 (38 feet deep).

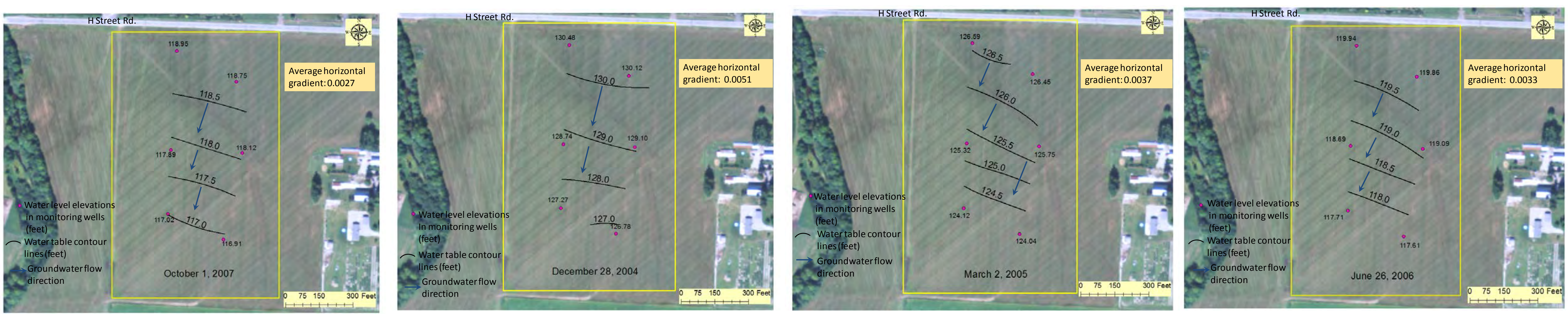


Figure 30. Water level contours in feet based on depth-to-water measurements on October 1, 2007 (fall), December 28, 2004 (winter), March 2, 2005 (spring), and June 26, 2006 (summer) and average horizontal gradient on each date. Elevations are relative to the top of casing at AKG721 (134.00 feet, NAVD88).

Plate 3. Water table elevations, depths, contours, and vertical hydraulic gradient.

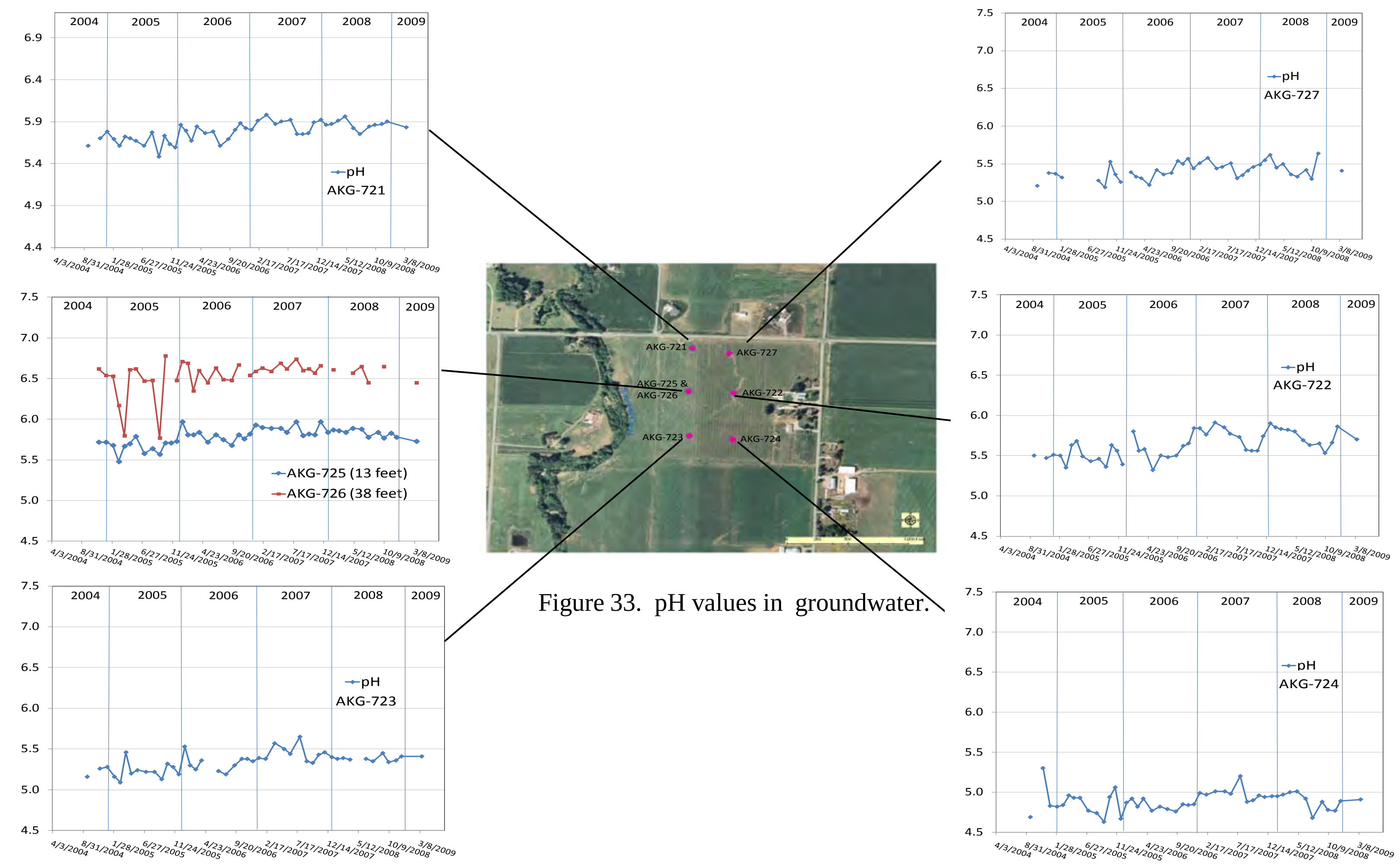


Figure 33. pH values in groundwater.

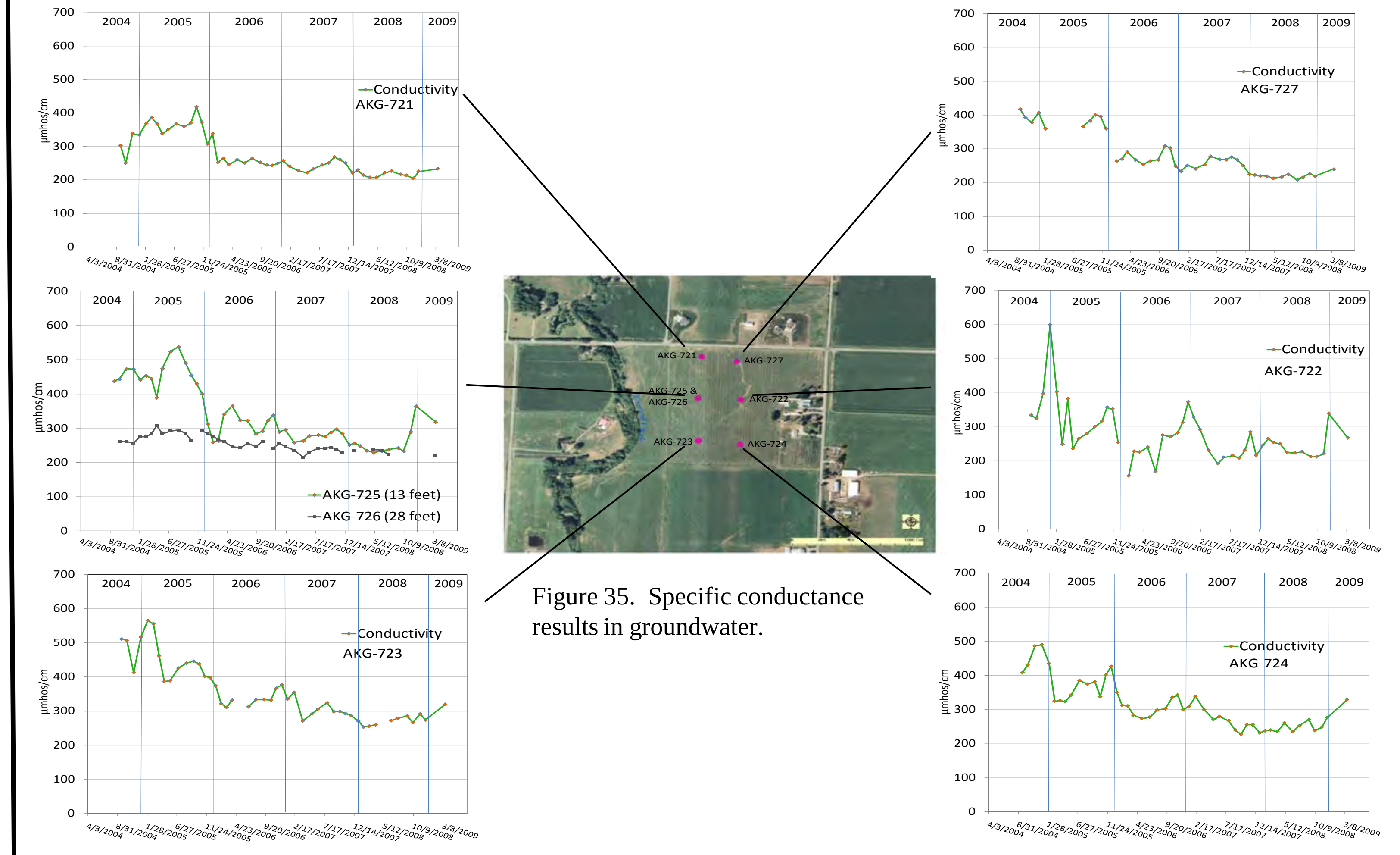


Figure 35. Specific conductance results in groundwater.

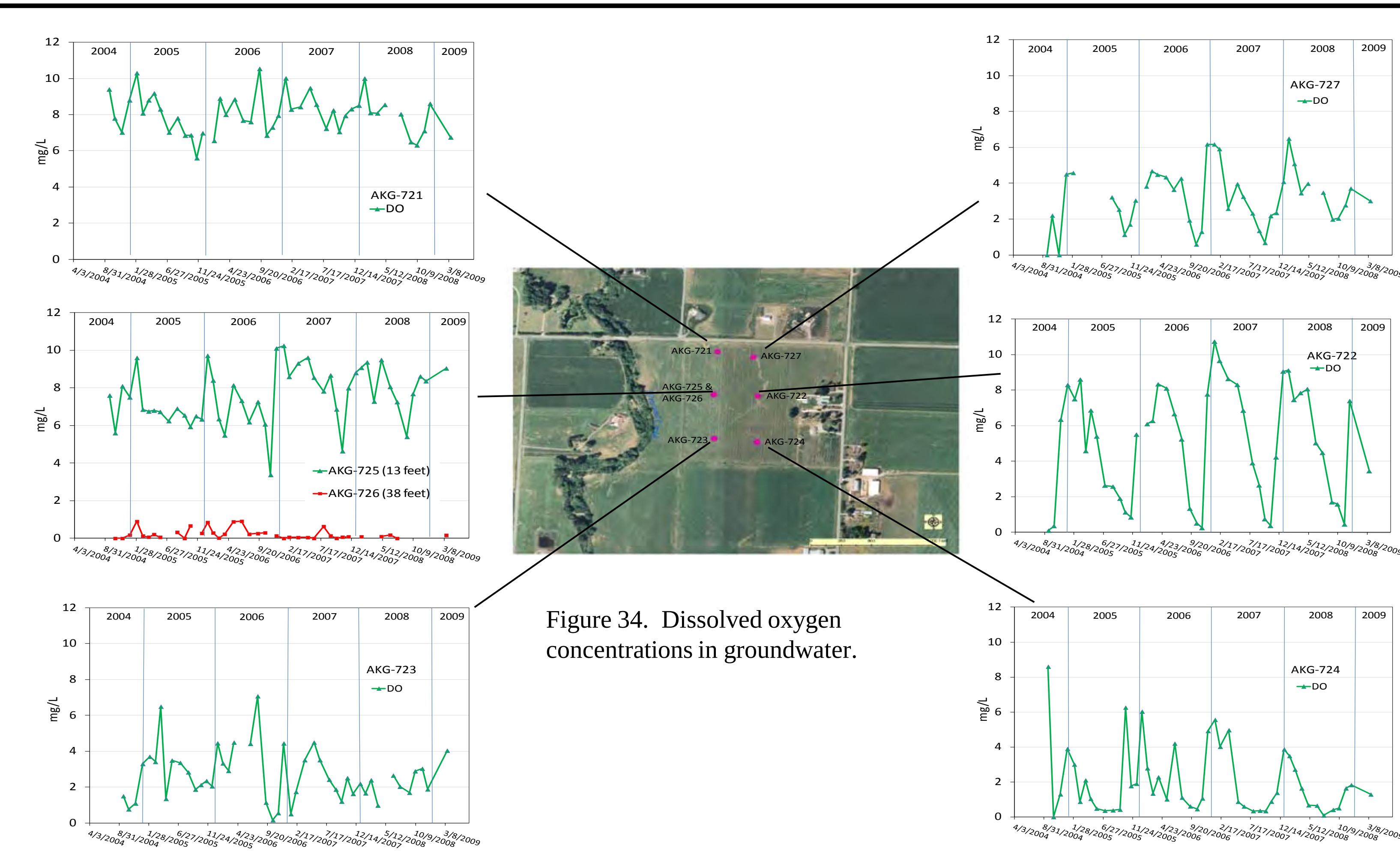


Figure 34. Dissolved oxygen concentrations in groundwater.

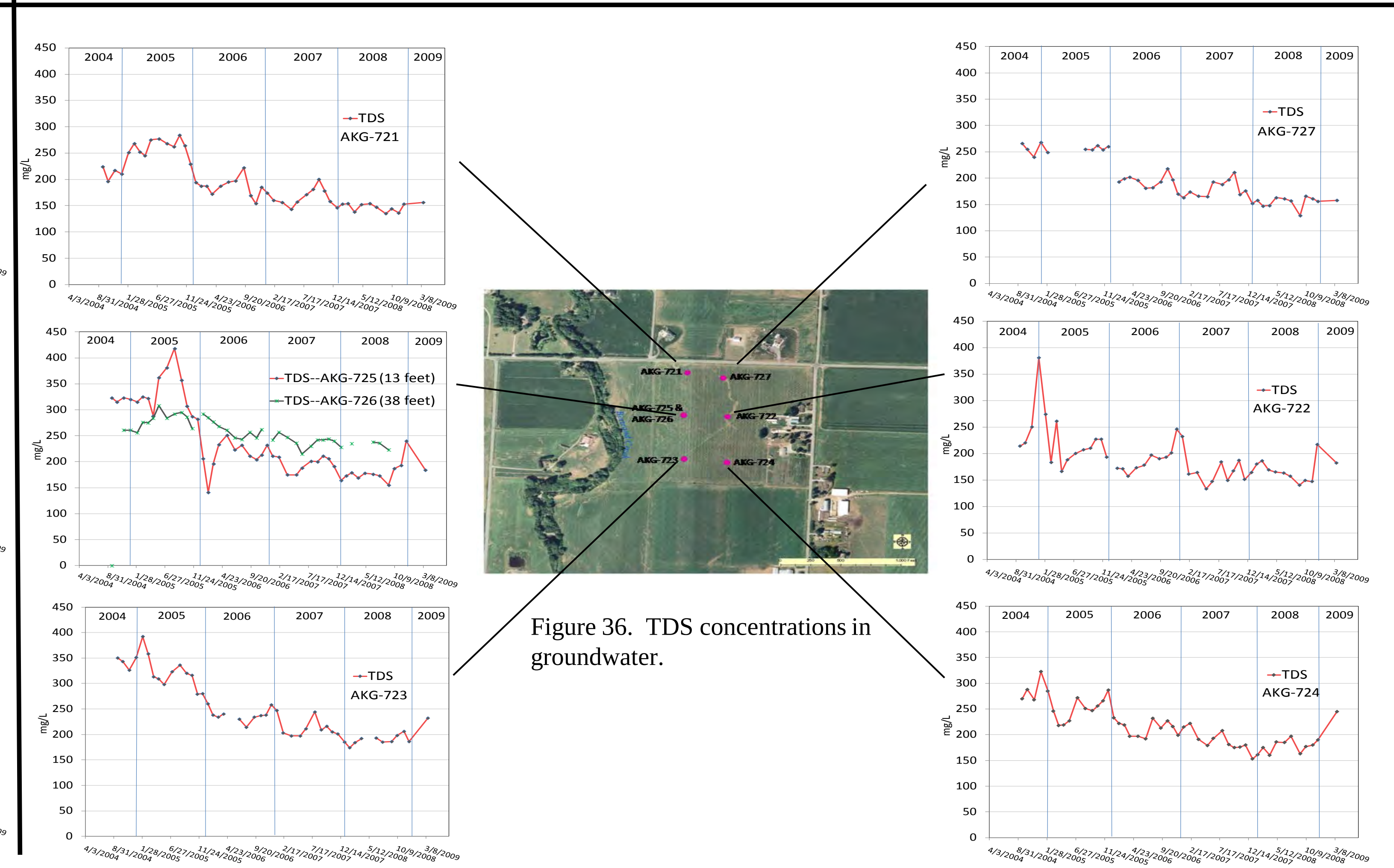


Figure 36. TDS concentrations in groundwater.

Plate 4. pH, dissolved oxygen (DO), specific conductance, and total dissolved solids (TDS) concentrations in groundwater. Concentrations are in mg/L except specific conductance in umhos/cm.

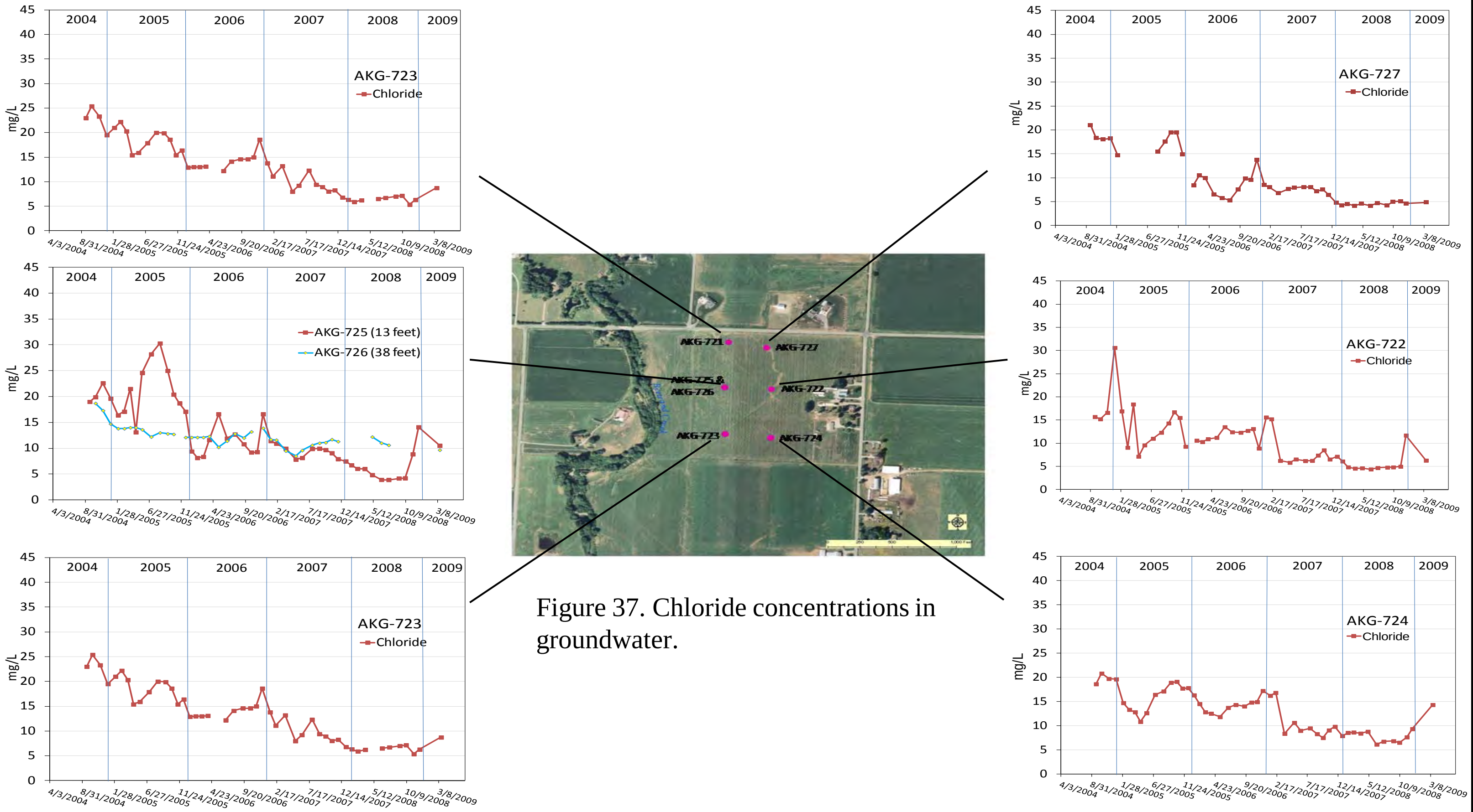


Figure 37. Chloride concentrations in groundwater.

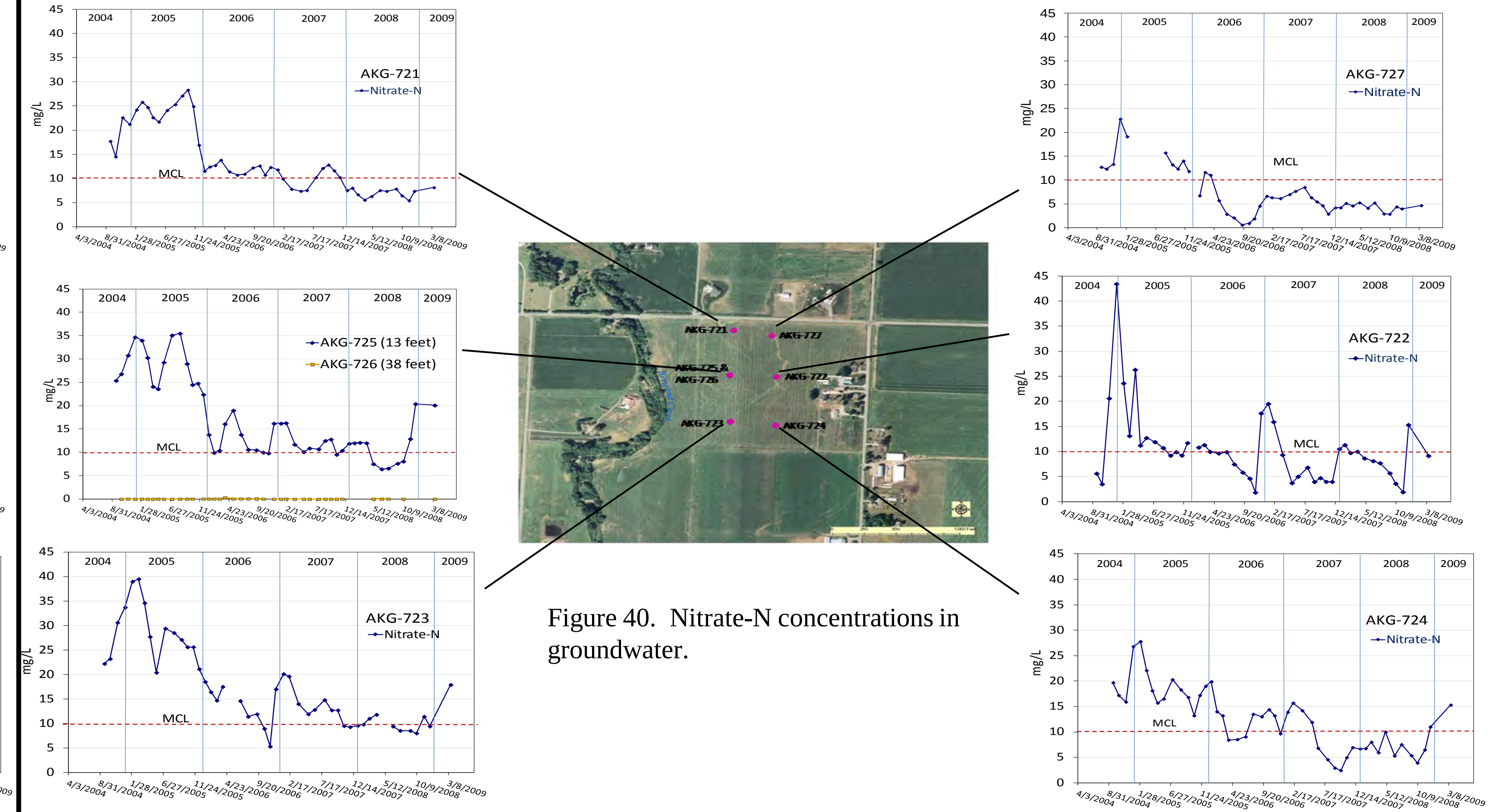


Figure 40. Nitrate-N concentrations in groundwater.

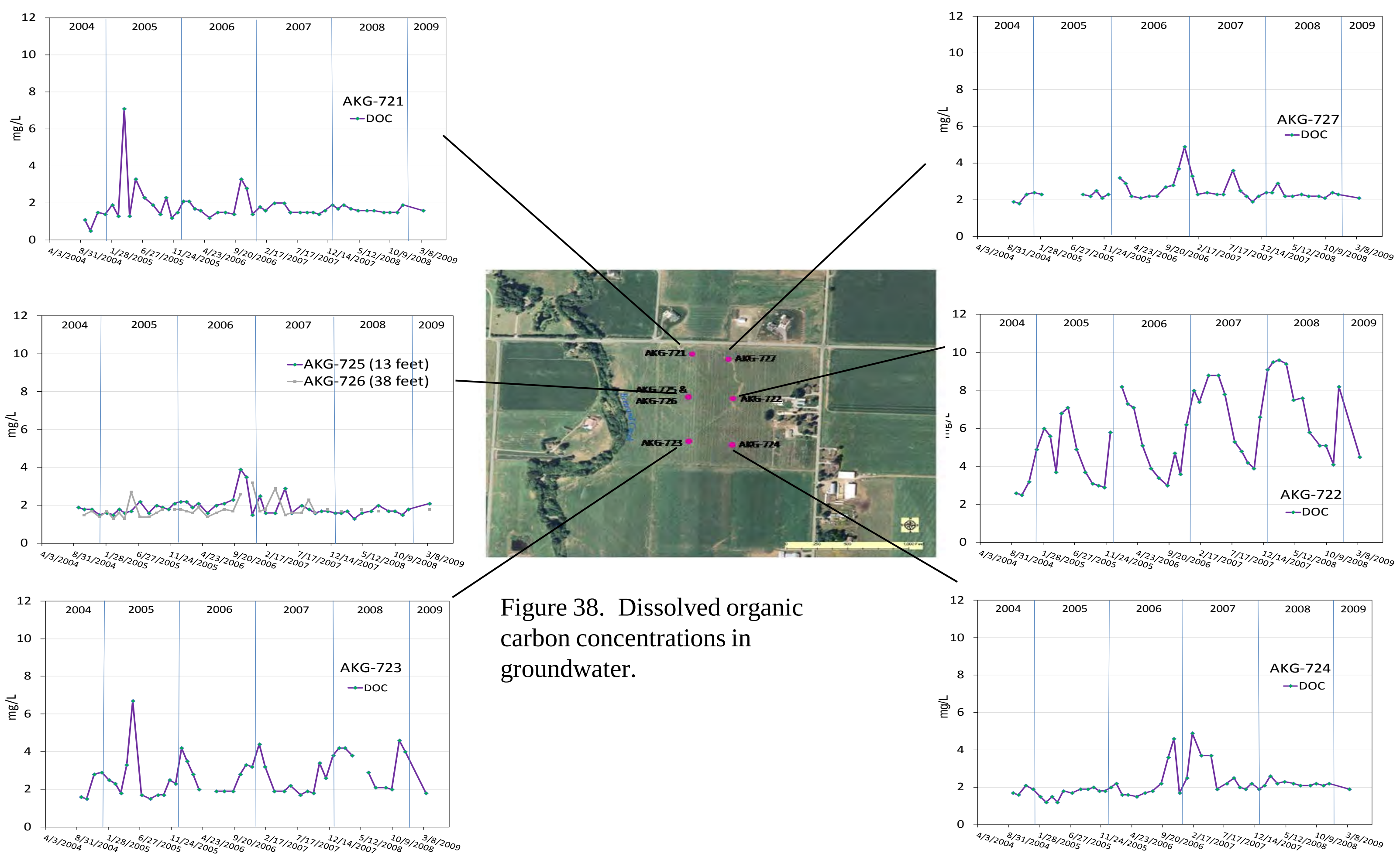


Figure 38. Dissolved organic carbon concentrations in groundwater.

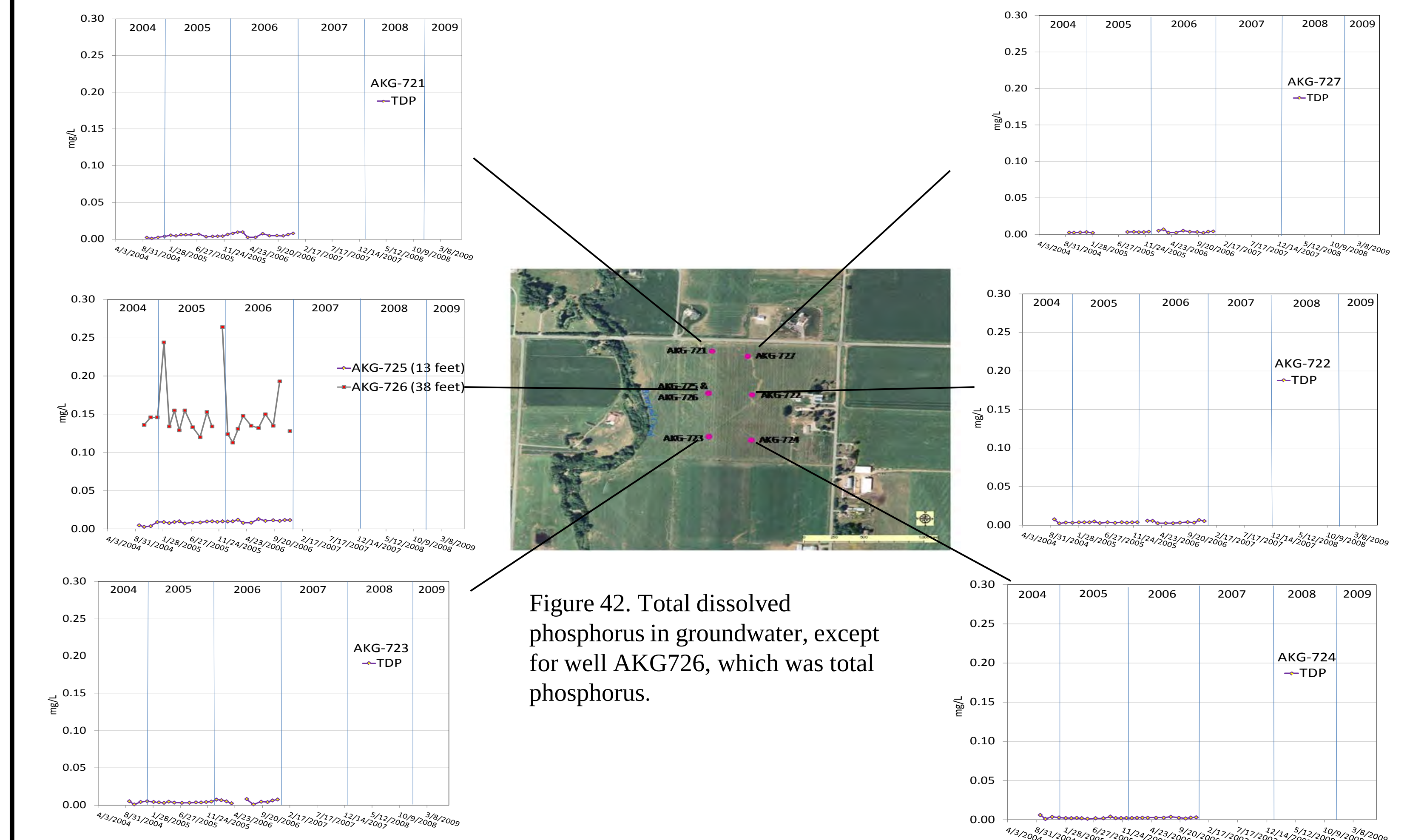


Figure 42. Total dissolved phosphorus in groundwater, except for well AKG726, which was total phosphorus.

Plate 5. Chloride, dissolved organic carbon, nitrate-N, and total dissolved phosphorus concentrations (mg/L) in groundwater.

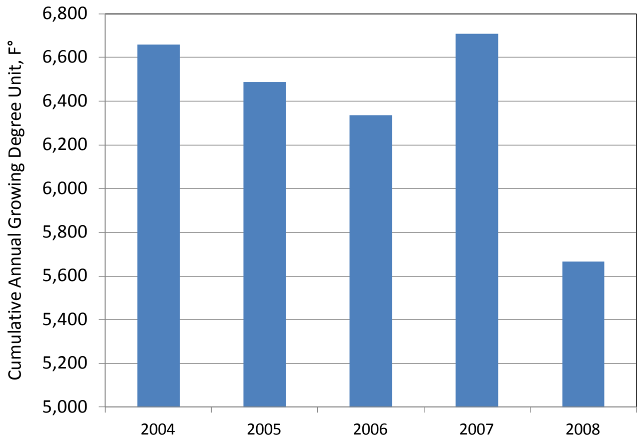


Figure 45. Annual total growing degree days from VanWieringen and Harrison (2009).

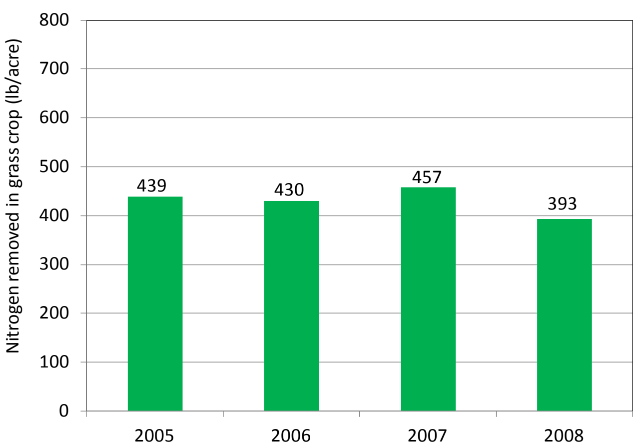


Figure 46. Annual nitrogen mass removed in the grass crop.

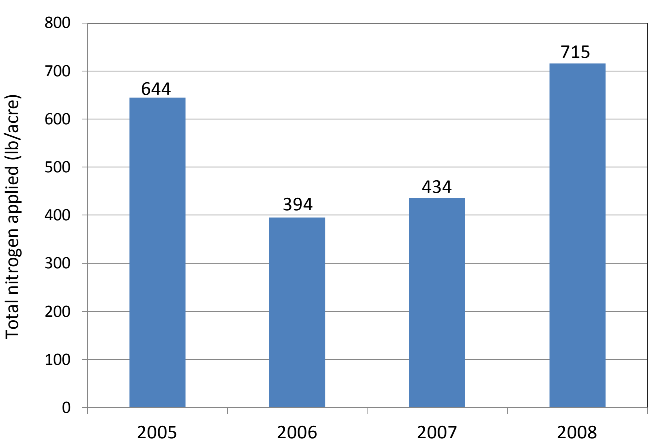


Figure 47. Total nitrogen applied annually (manure, fertilizer, and irrigation water).

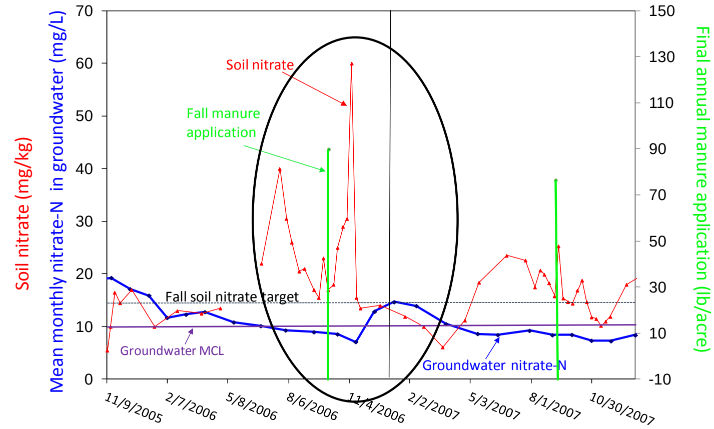


Figure 51. Soil nitrate and mean shallow groundwater nitrate-N concentrations. The circle points out the results of manure application on October 5, 2006.

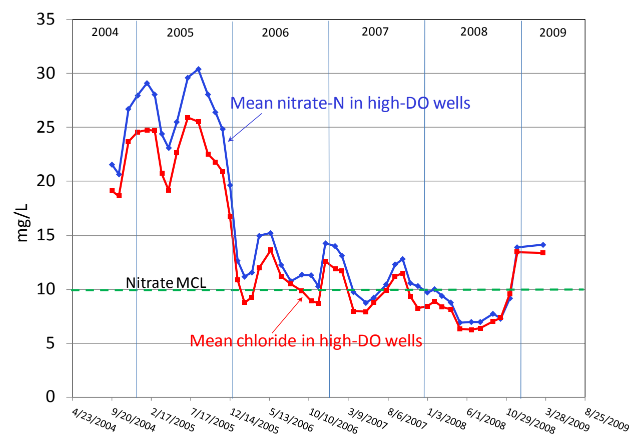


Figure 52. Nitrate-N and chloride concentrations in wells with high dissolved oxygen (DO).

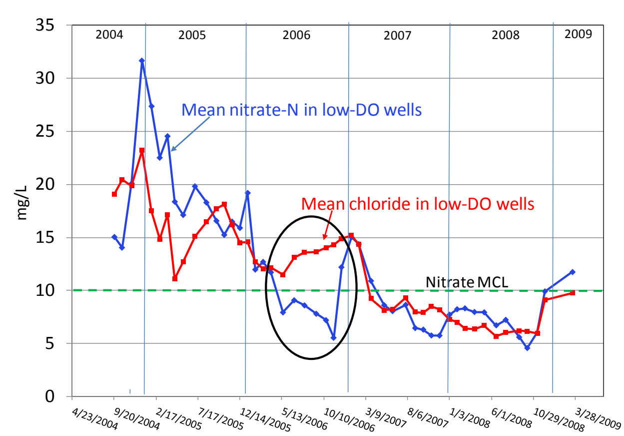
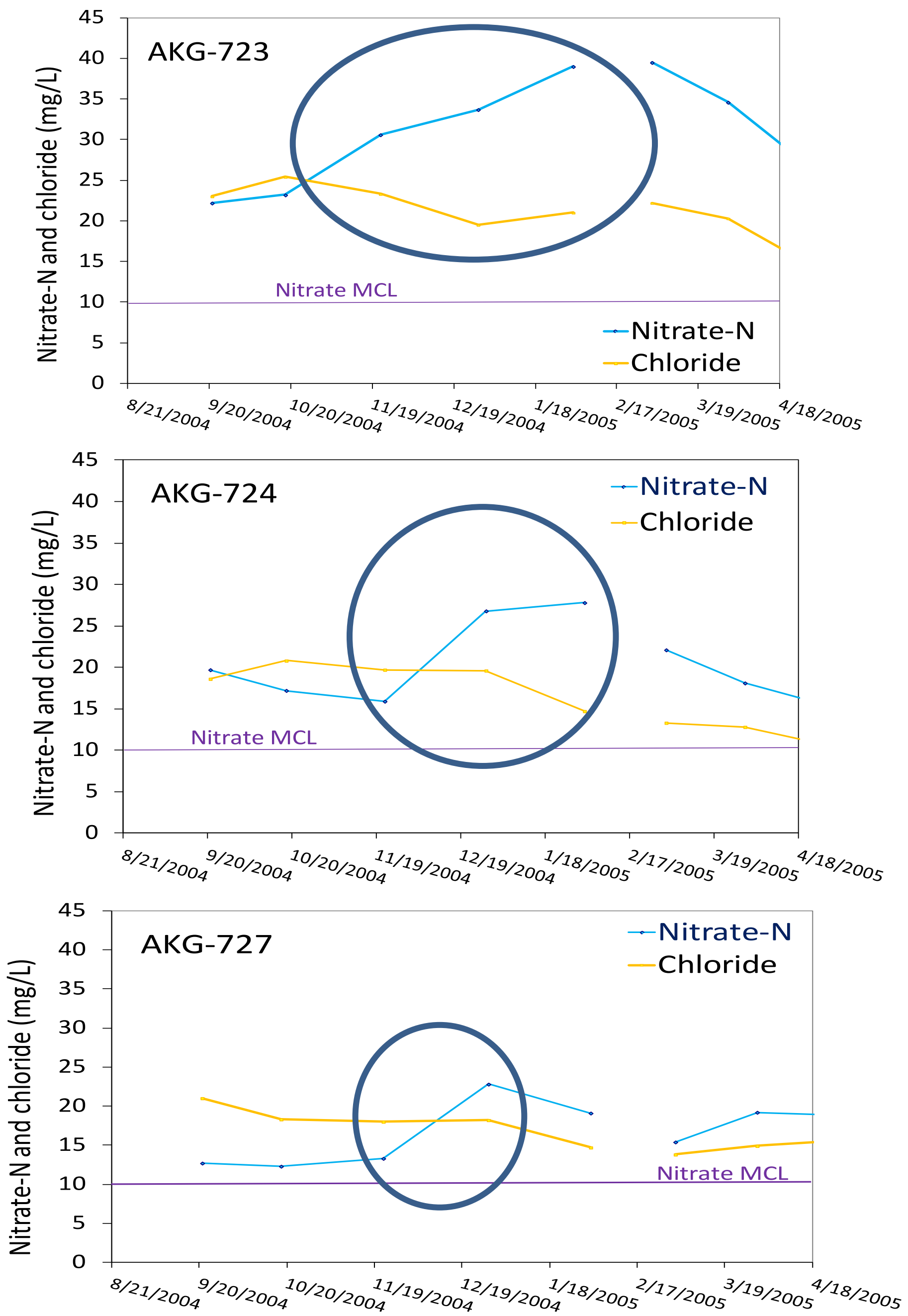


Figure 53. Nitrate-N and chloride concentrations in wells with low DO. The circle indicates a period when denitrification was probably a strong influence, because DO was less than 2 mg/L, and the water table below the root zone.

Plate 6. Discussion figures.

Winter 2004 to 2005



Winter 2007 to 2008

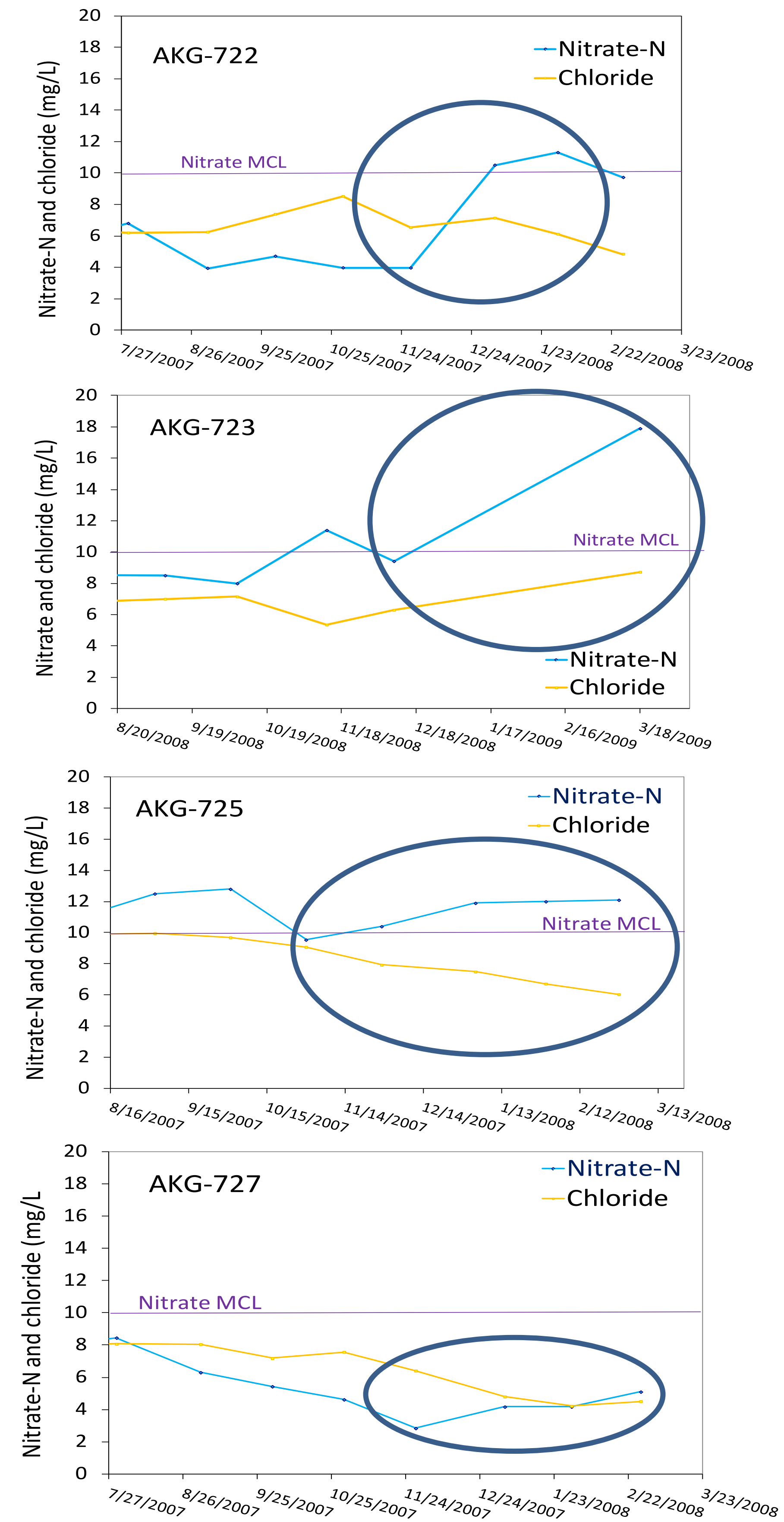


Plate 7. Nitrate and chloride concentrations during the winters of 2004-2005 and 2007-2008 in shallow groundwater. Circles indicate times in the winter when nitrate increases and chloride does not.

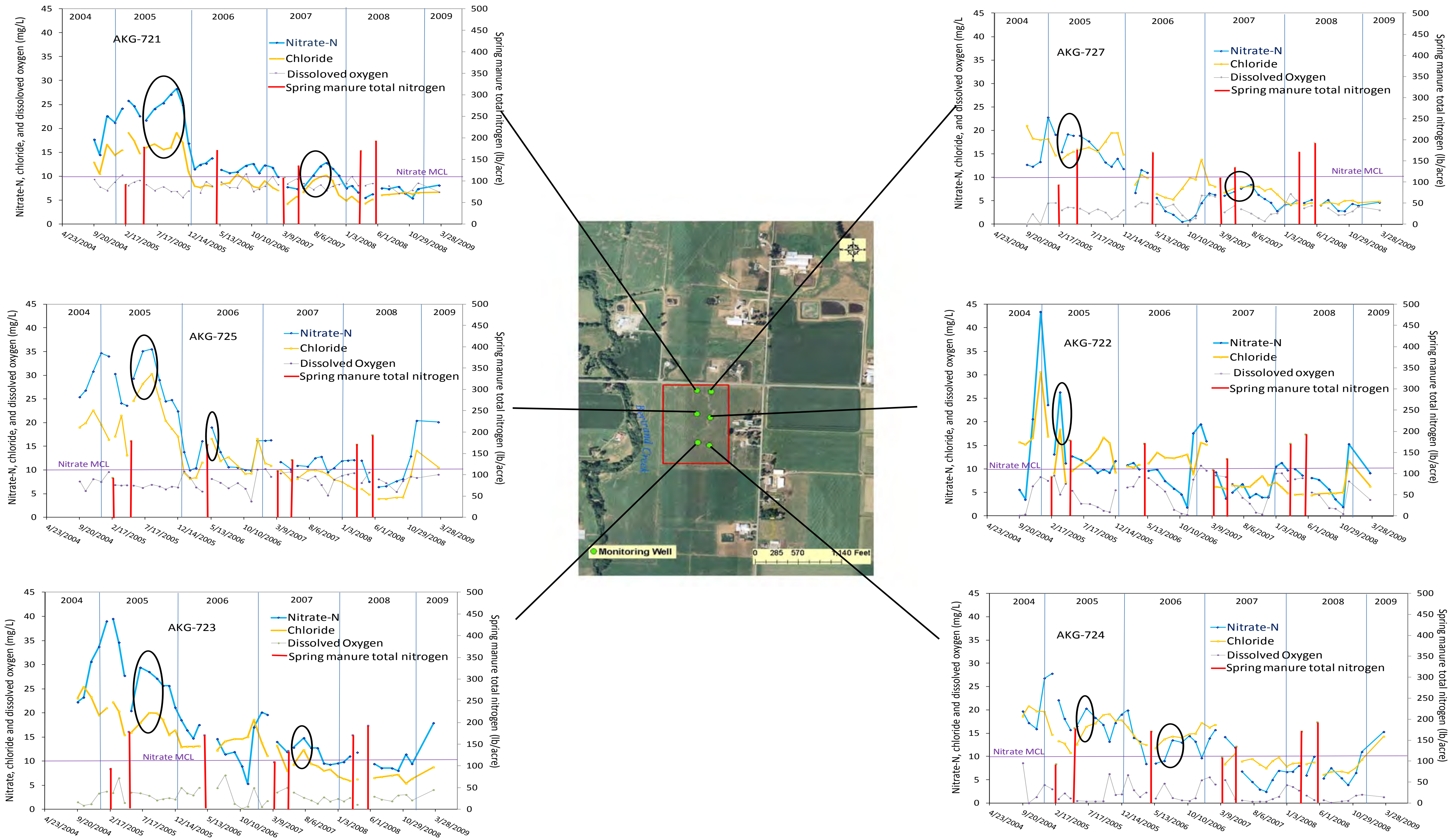


Plate 8. Nitrate-N, chloride and dissolved oxygen concentrations in individual monitoring wells in mg/L and spring manure total nitrogen applied in lb/acre. Circles show times when spring manure application was followed by elevated groundwater nitrate concentrations.

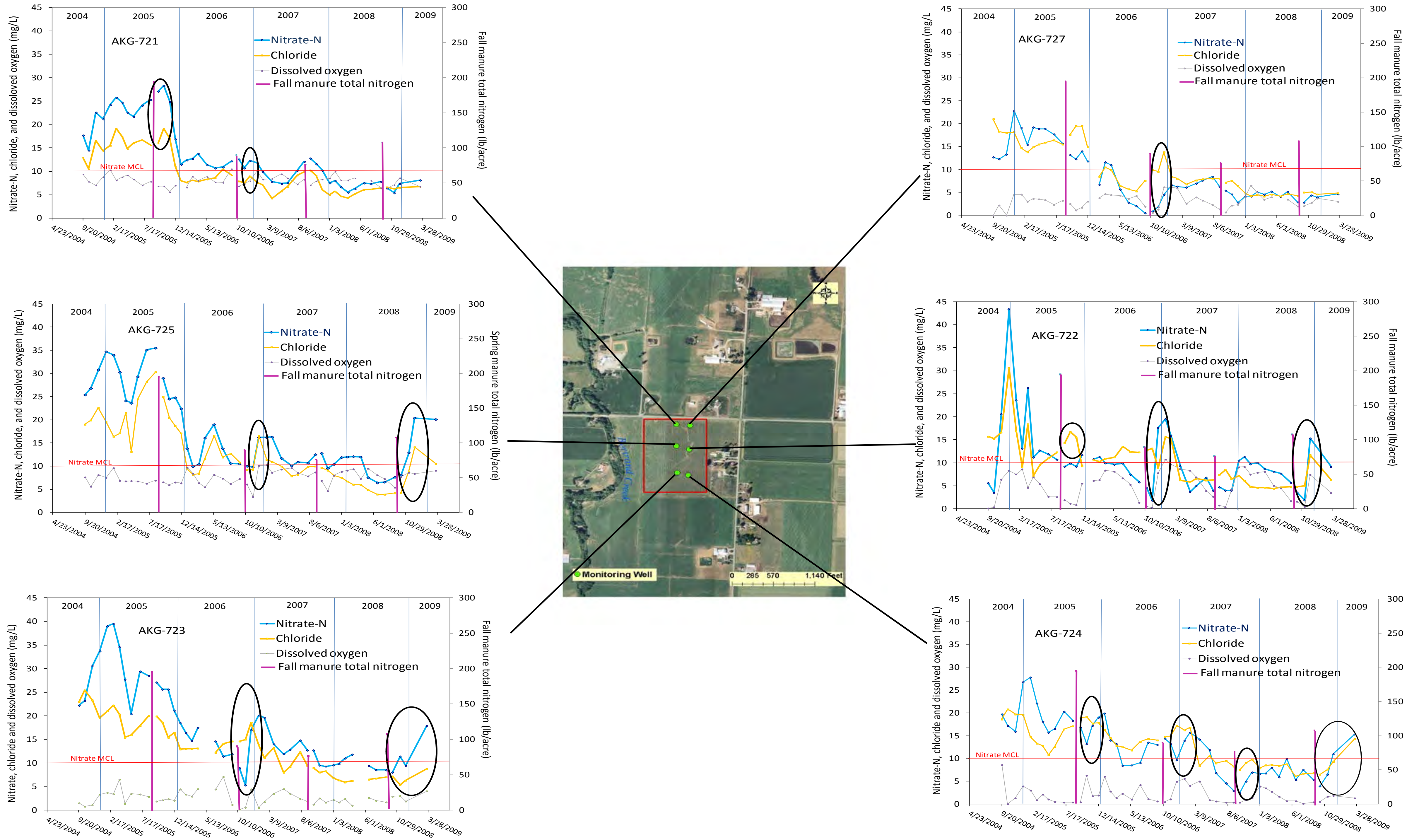


Plate 9. Nitrate, chloride and DO concentrations and total nitrogen application in fall 2005 through 2008. Circles indicate instances when chloride and nitrate increase following fall manure applications.